ANTHROPOGENIC IMPACTS ON POLAR BEAR BIOLOGY AND THE ARCTIC ECOSYSTEM

A Professional Paper

by

John E. Jordan

Submitted to the Office of Graduate Studies of Texas A&M University in partial fulfillment of the requirements for the degree of

Master of Wildlife Science

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Approved by:
Chair of Committee:
Committee Members:

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ABSTRACT

Anthropogenic Impacts on Polar Bear Biology and the Arctic Ecosystem.

(October 2013)

John E. Jordan, B.S., Texas A&M University

Chair of Advisory Committee: Dr. Miguel Mora

Despite its relative distance from most populated regions of the world, the Arctic has been significantly impacted by anthropogenic contamination and climate change. The entire Arctic ecosystem has been affected, with upper trophic level predators such as polar bears (*Ursus maritimus*), ultimately experiencing the worst impacts. The vast group of contaminants classified as persistent organic pollutants (POPs) are the primary anthropogenic source of Arctic contamination. Temporal trends indicate some legacy contaminants such as DDT have been steadily decreasing. However, data reflect that perfluorinated carboxylic acid (PFCA), BFRs, and other substances are generally increasing in concentration among most polar bear populations. For most POPs, the highest concentrations are found among the East Greenland and Svalbard populations, with the lowest concentrations found in the Alaska population. Exposure to some POPs can reduce vitamin concentrations in tissue and blood, affect the endocrine system by altering thyroid, cortisol, and reproductive hormone synthesis and transport, alter the development and function of sexual organs, induce immunosuppression, damage internal organs, reduce bone mineral density, and alter behavior. Numerous new chemicals which have not been previously found in the Arctic and in polar bears are beginning to show up in tests of polar bear tissue samples. Concentrations of most metals are highest among western bear populations such as those in Alaska, with levels decreasing in an easterly direction. In polar bears, mercury is known to damage internal organs, the neurological system, and the endocrine system. Climate change is already impacting polar bears due to changes in sea ice and other factors and these effects are expected to worsen in the coming years. The current collection of data regarding the health effects of anthropogenic contamination and climate change on polar bears certainly indicate the situation merits continued and expanded research.

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INTRODUCTION

Anthropogenic contamination in the Arctic was first documented in the 1960s. Prior to this time, the Arctic was assumed to be virtually devoid of anthropogenic contaminants since Arctic regions are characterized as having historically low population densities and minimal prevalence of industrial and agricultural operations. Sadly, as the decades passed, increasingly more evidence surfaced as to the extent of contaminants which have reached the Arctic via atmospheric and oceanic transport and other means from lower latitudes (Corsolini et al., 2002).

Contaminants vary in environmental significance, with the contaminants posing the most risk characterized as being persistent in the environment, difficult for organisms to metabolize and/or excrete, and easily transferred between trophic levels (Bard, 1999). Unfortunately, these characteristics apply to many of the contaminants in the Arctic including substances such as persistent organic pollutants (POPs), heavy metals, and others (Brunström and Halldin, 2000). Studies have detected contaminants in Arctic air, snow, freshwater, freshwater organisms, seawater, fog, ice, aquatic sediment, soil, plants, terrestrial organisms, and marine organisms. Together with contaminants, anthropogenically-influenced climate change also causes major negative impacts on the Arctic ecosystem (Bard, 1999).

According to the Stockholm Convention, POPs are chemicals discovered in locations distant from sources of release and for which long-range transport likely occurred (de Wit et al., 2006). In recent decades, international regulations have managed to reduce the use and subsequent continued Arctic accumulation of some legacy contaminants. However, levels of other legacy contaminants have been slow to respond and new contaminants are actually rapidly increasing (McKinney et al., 2011).

Polar bears (*Ursus maritimus*) are especially vulnerable to accumulation of contaminants and therefore are a reliable indicator species with which to gauge the extent of Arctic ecosystem contamination (Fisk et al., 2005). In fact, they are one of the most contaminated species in the Arctic

(Braune et al., 2005). This is largely due to their position at the top of the food chain and intake of vast quantities of lipids such as seal blubber which typically contain the highest levels of contaminants of all body tissues (Fisk et al., 2005). Life spans generally exceed 20 years. On average, females have two cubs every third year, which is a rather low reproductive rate relative to other organisms and contributes to the low population growth rates of polar bears (Sonne, 2010).

Conditions in the Arctic, particularly with respect to sea ice, change drastically between summer and winter and prompt most subpopulations of polar bears to travel vast distances in search of food. Polar bears have adapted by undergoing periods of fasting when prey is scarce or difficult to acquire and periods of intense feeding when prey is plenty. However, since many contaminants are stored in fat, during periods of fasting when fat is lost, these fat-associated contaminants are released into the blood and other tissues (Brunström and Halldin, 2000). Due to the low reproductive rates of polar bears and seasonal fluctuations in fat content, contaminants which impact the endocrine system, immune system, internal organs such as the kidneys and liver, and other body systems not only threaten the health of individuals, they threaten the sustenance of entire populations (Sonne, 2010).

In 2008, after thorough analysis and repeated delays, the U.S. Fish and Wildlife Service declared the polar bear a threatened species. A factor alluded to in the decision was the steadily retreating sea ice as a result of climate change and the effects it has had and is expected to have in the future on the circumpolar polar bear population. While this listing is a step in the right direction for polar bear conservation, more action needs to be taken (Courtland, 2008). There are currently approximately 19 distinct subpopulations of polar bears, totaling an estimated 21,500-25,000 individuals, spanning a massive area which encompasses a variety of habitats as well as national jurisdictions (Derocher et al., 2004). Environmental regulations vary drastically between polar countries. Additionally, impacts from climate change will vary spatially and temporally in the Arctic. The combination of these factors will create challenges for coordinating and implementing polar bear conservation on a circumpolar-nation scale (Courtland, 2008).

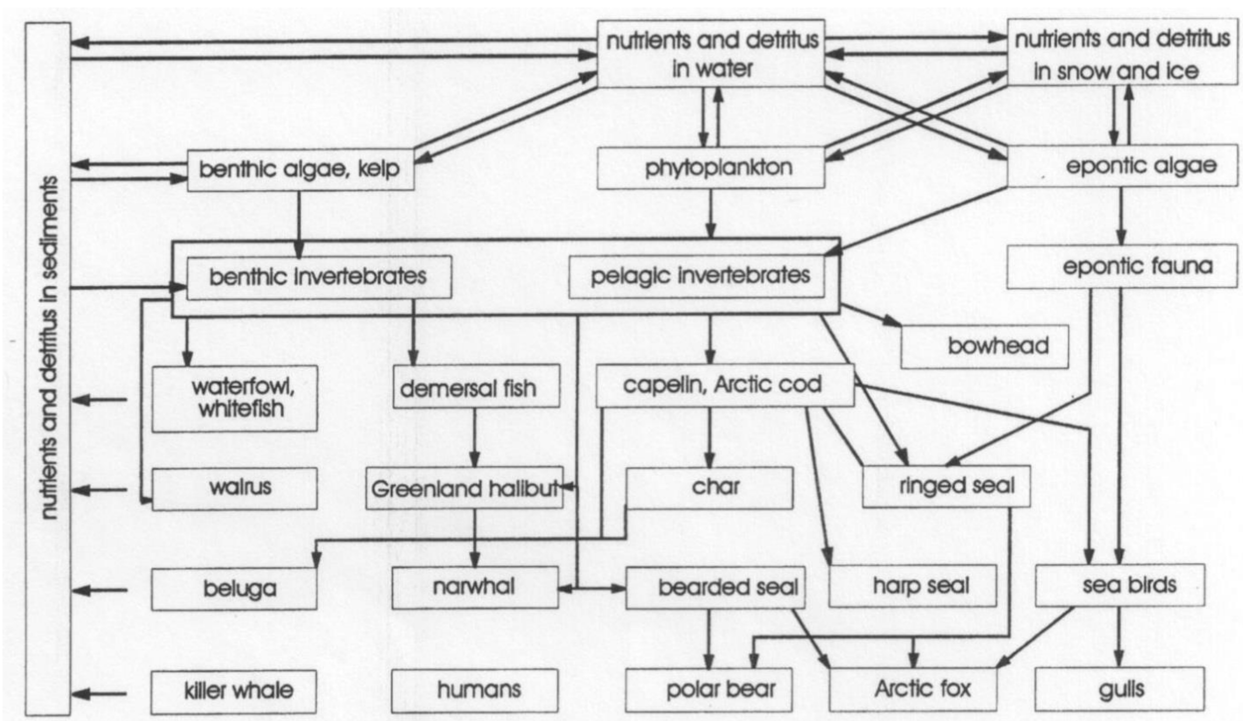
Objectives

Polar bears are an indicator species in the Arctic. As such, changes in the abundance or distribution of polar bears will be reflective of a changing Arctic ecosystem. In addition, there are intrinsic values to the sustained presence of polar bears in the Arctic which cannot be measured in ecological or economic terms. This manuscript intends to elucidate the current and developing threats to polar bear health and overall population abundance. Contaminants enter the Arctic ecosystem due to both long-range transport mechanisms, as well as local inputs. These contaminants are relatively efficiently transferred between trophic levels in the Arctic food web as a result of various biological and environmental factors. POPs and heavy metals are the primary types of contaminants in the Arctic and vary in concentration both spatially and temporally in the environment and in the biological tissues of various species. Both groups of contaminants are capable of bioaccumulating and biomagnifying up the food chain, ultimately causing a plethora of biological impacts to contaminated species. New chemicals, of which little is known with respect to potential biological effects, are routinely discovered in the Arctic ecosystem and biota. Additionally, the effects of climate change threaten the Arctic ecosystem and endemic species such as the polar bear and compound the biological impacts of contaminants. Furthermore, a secondary objective of the manuscript is to convey the unfortunate reality that the aforementioned threats are almost entirely caused by the actions of mankind, and subsequently the solutions will need to be effected by mankind. By compiling a relatively comprehensive collection of the causes and effects of various anthropogenic substances or actions, along with similar informative efforts by other concerned individuals, perhaps there remains future hope for more responsible stewardship of the Arctic ecosystem and its endemic biota, most notably the polar bear.

FOOD WEB

Due to a variety of factors such as abiotic conditions, low levels of nutrient input, a short growing season, and a short evolutionary history, the unique ecosystem of the Arctic is characterized as having low primary productivity, low biodiversity, and low population densities as compared to most ecosystems in lower latitudes (Bard, 1999). Primary productivity in the Arctic is driven by nutrient inputs from rivers, upwelling from southern seas, and atmospheric deposition. Relatively simple predator-prey relationships comprise the Arctic food web (Figure 1) (Muir et al., 1999).

Figure 1. Arctic marine food web. (Muir et al., 1999)



For example, a very general Arctic food web in order of increasing trophic level is as follows: plankton-amphipods-fish-marine mammals such as seals-polar bears (Bard, 1999). Most Arctic animals store large amounts of lipids which are used as an energy source to counter the effects of the cold temperatures. As such, these lipids are transferred throughout the food chain from one trophic level to another (Brunström and Halldin, 2000).

Polar bears are specialized but opportunistic predators of the highest trophic level (Corsolini et al., 2002). Their diet varies slightly between regions, dependent upon the prevalence of potential prey species in each respective region (Kucklick et al., 2002). They feed primarily on marine mammals such as seals (*Pusa hispida* and *Erignathus barbatus*), walrus (*Odobenus rosmarus*), bowhead (*Balaena mysticetus*), and beluga (*Delphinapterus leucas*), but will scavenge carrion if presented an opportunity (Woshner et al., 2001). When feeding, polar bears preferentially consume blubber and skin, both rich sources of lipids. Interestingly, even when unfavorable ice conditions force polar bears onto land, they do not consume significant quantities of terrestrial food sources (Muir et al., 1999).

Many of the species which polar bears prey upon utilize differing trophic pathways. Ringed seals consume mostly fish, amphipods, and euphausids. Beluga whales have a similar diet. In contrast, the diets of bearded seals and walrus consist of shrimp, crabs, mollusks, and occasionally fish. Bowhead whales feed on euphausids and copepods (Kucklick et al., 2002).

Changes in population density or abundance among species at any trophic level can alter the entire trophic structure of the food web with dramatic consequences on the populations of other species. For example, bottom-up change may occur if primary productivity is affected by changes in factors such as nutrient levels or light intensity. The effects reverberate upward through the food chain, and ultimately, higher trophic level animals may be shifted in the trophic structure. This type of change has been documented in the Bering Sea and Arctic Ocean. In contrast, top-down change is a result of external factors affecting the populations of higher trophic level animals, consequently impacting populations of animals at lower trophic levels (Macdonald et al., 2003).

TRANSPORT MECHANISMS

Contaminants are transported to and within the Arctic by various means. The primary pathways are atmospheric transport, transport via rivers, and transport by ocean currents (Figures 2 and 3) (Muir et al., 1999; Macdonald et al., 2003).

Figure 2. Transport, deposition, redistribution, and accumulation of POPs and metals. (Muir et al., 1999)

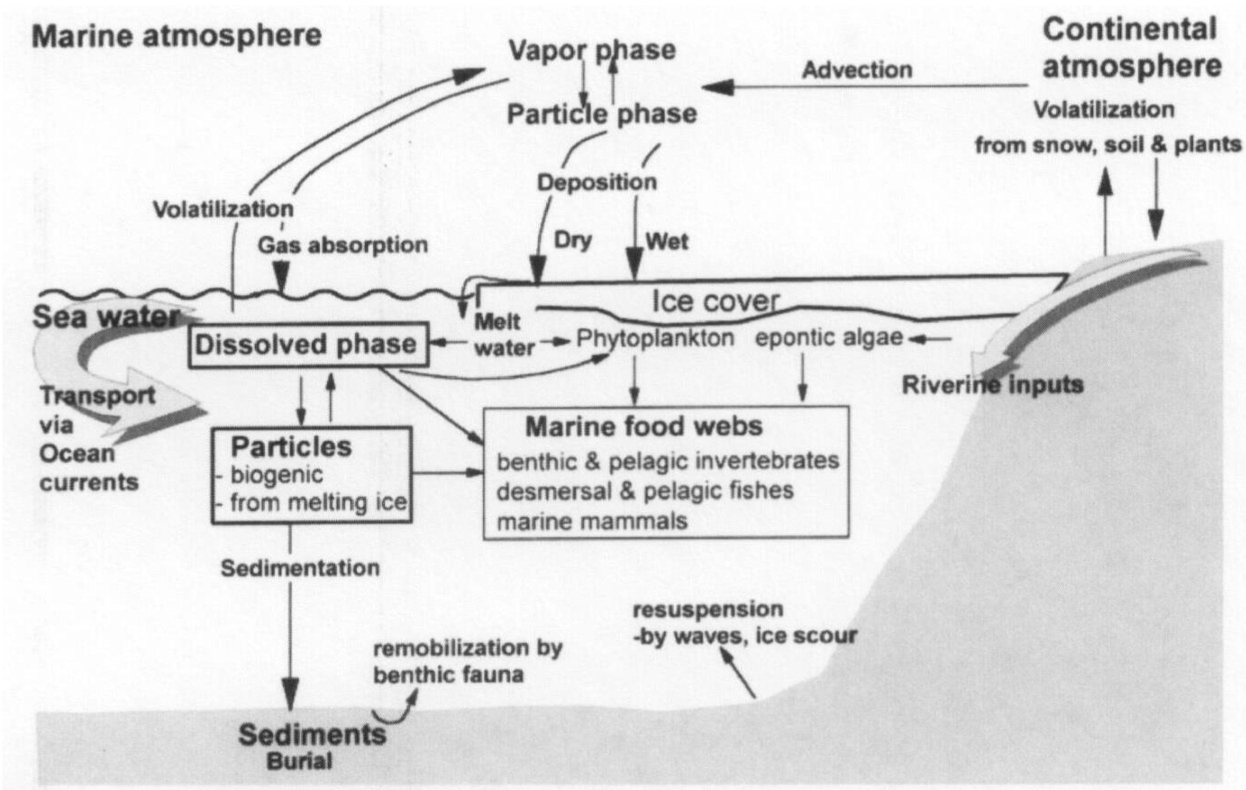
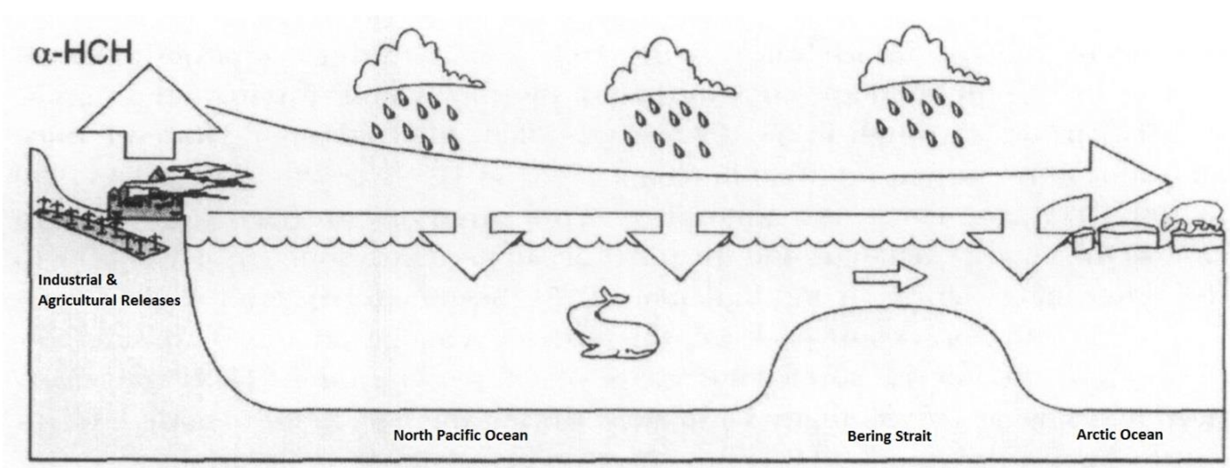


Figure 3. Atmospheric and oceanic transport. (Macdonald et al., 2003)



Atmospheric transport is influenced by seasonal weather-related fluctuations. For example, winter in the northern hemisphere generally consists of persistent low pressure systems present over the northern Pacific and Atlantic Oceans and high pressure systems persistent at lower latitudes. In addition, high pressure systems situated in Siberia push air currents north into the Arctic which accounts for much of the Eurasian atmospheric contaminant input to the Arctic (Brunström and Halldin, 2000). Eurasian sources are responsible for over 50% of the air pollution in the Arctic (Bard, 1999). During this period of time, atmospheric transport of contaminants from lower latitudes into the Arctic, referred to as global distillation, is at its annual maximum (Brunström and Halldin, 2000). An annual occurrence referred to as Arctic haze results from this influx of atmospheric contaminants and is most evident from December through April. The haze consists of a plethora of contaminants including acidifying gases, soot, heavy metals, and POPs (Bard, 1999).

Most atmospherically transported contaminants are considered volatile substances (Braune et al., 2005). The volatile nature of some contaminants, coupled with warmer temperatures and higher rainfall common at low latitudes, causes the contaminants to easily enter the atmosphere due to evaporation from soils and water. The process whereby these volatile contaminants condense and concentrate in the Arctic has been termed global fractionation (Bard, 1999). Atmospheric transport may occur with contaminants in the form of a gas or with contaminants absorbed to particles or suspended in the air. Gas absorption by water, in the form of rain or snow, provides a route for transfer of gaseous contaminants to ground level. Precipitation scavenging involves the removal of particulate contaminants by rain or snow from the atmosphere, ultimately depositing the contaminants at ground level. Dry deposition of particulate contaminants, absent the need for absorption by rain or snow, is another means for atmospheric contaminants to reach ground level (Braune et al., 2005).

Many POPs, including brominated flame retardants (BFRs), polychlorinated naphthalene (PCN), and other chemicals with similar physical properties, are atmospherically transported to the Arctic from densely-populated areas at significantly lower latitudes in stages of subsequent deposition and revolatilization, rather than through one single event or a one-hop pathway (de Wit et al., 2006). Fluctuating thermal cycles and other processes that affect a chemical's vapor pressure drive the aforementioned revolatilization (Macdonald et al., 2003). This has been described as "the grasshopper effect" or multi-hop pathway and illustrates how these contaminants travel extremely long distances and arrive in remote, uninhabited areas of the Arctic (de Wit et al., 2006). Atmospheric transport from source of release to the Arctic occurs rather quickly, usually in days or weeks, regardless of geographic origin.

In contrast, transport by ocean currents is a significantly slower process, usually measured in years (Macdonald et al., 2003). Most contaminants transported via rivers or ocean currents are less volatile and more water soluble than the atmospherically-transported contaminants (Braune et al., 2005). Although when transported by water, dichlorodiphenyltrichloroethane (DDT) and polychlorinated biphenyls (PCBs) are generally bound to particulate matter rather than dissolved in the water column (Muir et al., 1999). North-flowing rivers in Eurasia and North America drain approximately 10,000,000 km^2 of surface land area into the Arctic, which contributes significant contamination due to direct and indirect upstream releases into the respective rivers (Brunström and Halldin, 2000).

Although of minimal relative contribution to Arctic contamination, migratory animals such as whales can transport contaminants from lower latitudes and deposit them via excrement or tissue decomposition or consumption by scavengers, upon death (Corsolini et al., 2002). Within the Arctic, additional contaminant sources include accidental spills, intentional releases, and combustion of wastes from facilities such as petroleum extraction operations, military radar installations, and municipal waste dumps. These contaminants are transported within the Arctic via atmospheric and water current pathways (Brunström and Halldin, 2000). Ice, such as that of drifting ice or permafrost, may trap contaminants. Upon melting due to seasonal temperature fluctuations or climate change, the contaminants will then reenter the environment at a later time and/or location (Corsolini et al., 2002).

PERSISTENT ORGANIC POLLUTANTS

The majority of persistent organic pollutants (POPs) are of anthropogenic origin via industrial activities and most have been produced only within the past five decades for use as pesticides, in consumer products, and in manufacturing processes (Braune et al., 2005). POPs vary with respect to their chemical properties and toxicities (Bard, 1999). The majority of POPs are resistant to physical, chemical, and biochemical degradation, easily stored in fat tissue, and biomagnify through the trophic levels, which explains their prevalence and high concentrations in numerous species and ecosystems throughout the world (Ropstad et al., 2006).

In both Arctic air and seawater, the five most prevalent POPs from highest to lowest concentration are as follows: hexachlorocyclohexane (HCH) (385-577 pg/m^3), toxaphene (36-44 pg/m^3), PCB (14-20 pg/m^3), chlordanes (4-6 pg/m^3), and DDT (1-2 pg/m^3) (Bard, 1999). PCBs were first reported to be found in polar bear tissue in 1975. Since then, a host of other POPs has been found in polar bear tissue such as DDT, HCH, chlorobornanes, chlordane, and many others (Muir et al., 1999). Additionally, it is important to mention that some families of POPs actually include numerous compounds. For instance, there are 209 different types of PCBs (Chiu et al., 2000).

Polar bears have the highest tissue concentrations of POPs in general of any animal in the Arctic. And due to a well-developed biotransformation capacity, they also have the highest concentrations of POP metabolites. Among polar bears, the accumulation, concentrations, and effects of POPs depend on several factors, including age and sex of the individual. For example, PCBs and DDT are usually lower in females, and this is believed to be due to loss of some of these contaminants by females via lactation. In contrast, males generally have lower levels of chlordane which is presumed to be due to males having a better capacity to metabolize this contaminant. Some POP concentrations, such as chlordanes, are higher in subadults, while others, such DDT, vary little between age groups (Braune et al., 2005).

In general across populations, the order of polar bear tissue concentrations of various POPs from highest to lowest is as follows: PCB, chlordane (CHL), perfluosulfonic acid (PFSA), DDT and metabolites, carboxybenzyl (CBz), HCH, toxaphene, perfluorinated carboxylic acid (PFCA), polybrominated diphenyl ether (PBDE), and hexabromocyclododecane (HBCD). This list includes the POPs of highest concentrations in polar bears, although there are many more that have been detected in tissue in lower concentrations (Letcher et al., 2010). Furthermore, many of these chemicals have half-lives of 20 years or more in polar bear tissue (Sonne, 2010). Chlordane is the most acutely toxic compound in the list, but chronic exposure to any of the above-mentioned compounds is certain to cause biological damage (Bard, 1999). Given the protected status of the polar bear, most studies are performed using fat tissue or blood samples taken from chemically-immobilized individuals or those killed by subsistence hunters where permitted to do so. Conclusively, POPs are of significant biological risk to polar bear health, as they are known to affect development, hormone and vitamin levels, organs, bone density, neurological systems, immune systems, reproductive systems, and other body systems (Letcher et al., 2010). The toxic effects of these contaminants are magnified during periods of extreme stress such as fasting, breeding, lactation, and migration when fat tissue is undergoing dynamic changes and subsequently releasing stored contaminants into the blood stream (Sonne, 2010).

Spatial Trends

Spatial trend data for levels of POPs in polar bears, particularly legacy contaminants, are rather thorough. However, data are lacking for many of the newer contaminants such as certain BFRs, as well as for other POPs such as polychlorinated dibenzodioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), tris (p-chlorophenyl) methanol, photoheptachlor, and toxaphene. For reference, median (range) values for various POP concentrations from the entire Arctic polar bear population are as follows: PCB 7119 (1228-70,421) ng/g, CHL 1988 (207-15,013) ng/g, *p,p'*-DDE 190 (24-2821) ng/g, and dieldrin 149 (7-835) ng/g (Muir et al., 1999). Data indicate polar bear populations from areas with permanent sea ice typically have higher PCB levels than those populations from areas lacking permanent ice cover. This is believed to be due to differences in feeding ecology of ringed seals in the respective regions (Bard, 1999).

PCB and DDT levels are generally highest in East Greenland and Svalbard populations and decrease in a westerly direction, with the lowest levels among Alaskan populations (Table 1) (McKinney et al., 2011).

Table 1. Mean concentrations of PCB and DDT in adipose tissue of polar bears. (McKinney et al., 2011)

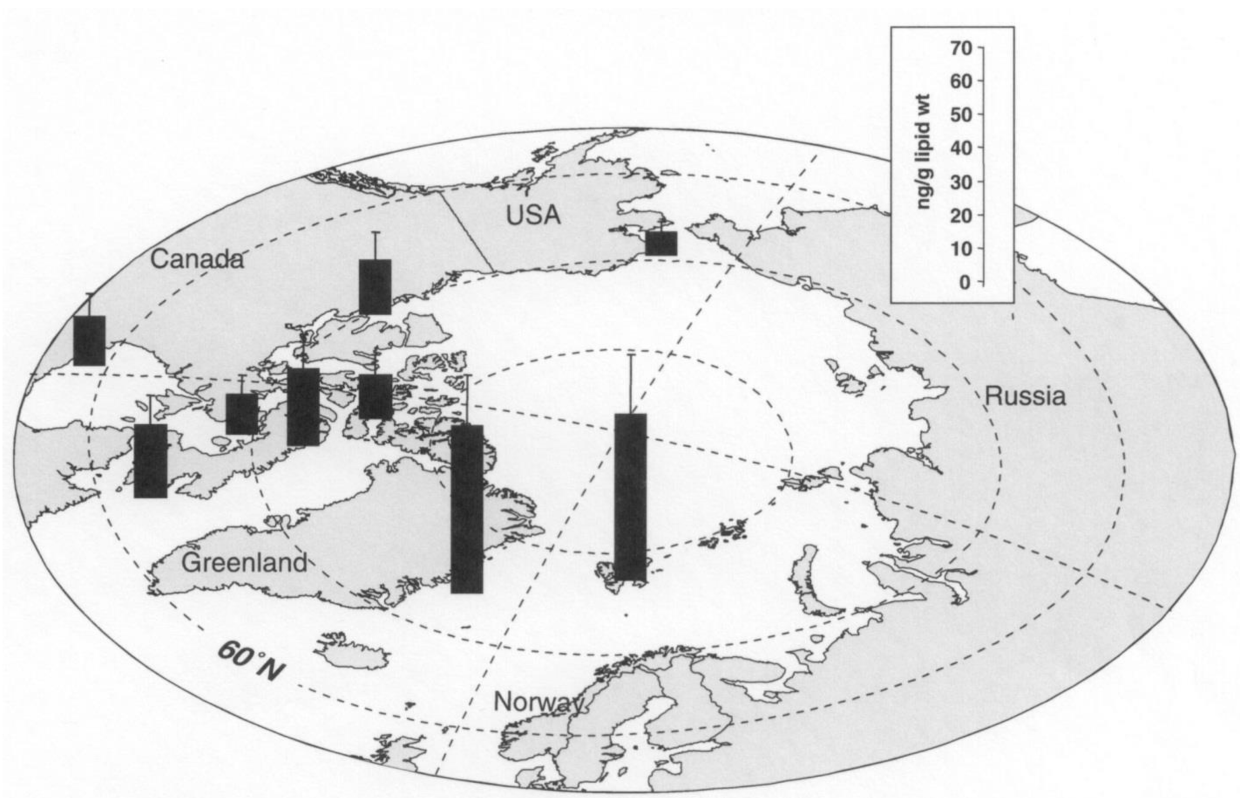
Location	PCB (ng/g lw)	DDT (ng/g lw)
Alaska	1842	77.2
S. Beaufort Sea	3688	81.8
N. Beaufort Sea	5541	93.7
Gulf of Boothia	2445	31.5
Lancaster/Jones Sound	2598	64.0
Baffin Bay	3211	179
Davis Strait	4674	104
W. Hudson Bay	4634	88.1
S. Hudson Bay	5523	152
E. Greenland	10,537	206
Svalbard	5137	119

East Greenland bears have mean PCB and DDT concentrations of 10,537 ng/g lw and 206 ng/g lw respectively, while mean PCB and DDT concentrations in bears from the Alaskan Arctic are 1842 ng/g lw and 77.2 ng/g lw respectively. In the Canadian Arctic, mean PCB concentrations range from 2445 ng/g lw in the Foxe Basin/Gulf of Boothia population to 5541 ng/g lw in the North Beaufort Sea population (McKinney et al., 2011). Cumulative data show PBDEs to be rapidly increasing in Arctic biota (Braune et al., 2005). The Svalbard population has a mean PBDE concentration of 50 ng/g lw. Populations in East Greenland and southeast Baffin have mean PBDE concentrations ranging from 22-70 ng/g lw. The lowest PBDE levels are found among populations from northwest Alaska, Foxe Basin, and Lancaster Sound, with mean concentrations ranging from 7-13 ng/g lw (Table 2 and Figure 4) (de Wit et al., 2006).

Table 2. Mean concentrations of PBDE in adipose tissue of polar bears. (de Wit et al., 2006)

Location	PBDE (ng/g lw)
Amundsen Gulf	16
Foxe Basin/Gulf of Boothia	7
Lancaster Sound	13
NE Baffin	19
SE Baffin	22
Western Hudson Bay	14
Svalbard	50
Bering/Chukchi Sea	6.7
Eastern Greenland	70

Figure 4. Mean concentrations of PBDE in adipose tissue of polar bears. (de Wit et al., 2006)



Ringed seal blubber from the aforementioned regions contained lower concentrations of PBDEs than were found in polar bear fat, thus indicating definite biomagnification (Braune et al., 2005). East Greenland polar bears receive an estimated average daily oral exposure of PBDE of 0.34 µg/kg body weight (Sonne, 2010). Trends in HBCD concentrations mirrored that of PBDE, with concentrations in the

European Arctic being higher than those in the North American Arctic. East Greenland and Svalbard populations have the highest levels, with mean concentrations ranging from 21.9-41.1 ng/g lw. The lowest HBCD levels are found among the Lancaster/Jones Sound population, with mean concentrations of 0.9 ng/g lw (Table 3) (McKinney et al., 2011).

Table 3. Mean concentrations of HBCD in adipose tissue of polar bears. (McKinney et al., 2011)

Location	HBCD (ng/g lw)
Alaska	-
S. Beaufort Sea	1.4
N. Beaufort Sea	1.8
Gulf of Boothia	-
Lancaster/Jones Sound	0.9
Baffin Bay	1.6
Davis Strait	2.9
W. Hudson Bay	4.2
S. Hudson Bay	5.2
E. Greenland	21.9
Svalbard	41.1

Given the spatial trends of PBDE and HBCD levels and atmospheric and ocean current patterns, source emissions of these contaminants most likely originate in eastern North America and Western Europe. PFCA distribution follows a similar pattern (Table 4) (Letcher et al., 2010).

Table 4. Mean concentrations of PFCA in adipose tissue of polar bears. (Letcher et al., 2010)

Location	Tissue	PFCA (ng/g ww)
East Greenland	Liver	500
Svalbard	Blood	320
Canadian Arctic	Liver	467
Southern Hudson Bay	Liver	515
Alaska	Liver	285

HCH concentrations are generally highest in the Alaskan Arctic populations (mean 490 ng/g lw) and decrease in an easterly direction, with populations in Svalbard having mean HCH concentrations approximately seven times lower than populations in Alaska (Table 5) (Letcher et al., 2010).

Table 5. Mean concentrations of HCH in adipose tissue of polar bears. (Letcher et al., 2010)

Location	HCH (ng/g lw)
East Greenland	200
Svalbard	71
Canadian Arctic	379
Alaska	490

This trend suggests HCH sources emanating from eastern Asia and North America (Verreault et al., 2005). Levels of chlordane concentrations are fairly evenly distributed and relatively high among all populations throughout the Arctic (Table 6) (McKinney et al., 2011).

Table 6. Mean concentrations of CHL in adipose tissue of polar bears. (McKinney et al., 2011)

Location	CHL (ng/g lw)
Alaska	765
S. Beaufort Sea	1268
N. Beaufort Sea	1982
Gulf of Boothia	1824
Lancaster/Jones Sound	1130
Baffin Bay	2167
Davis Strait	2135
W. Hudson Bay	3477
S. Hudson Bay	2166
E. Greenland	1732
Svalbard	1196

Concentrations of CBz, dieldrin, and mirex follow a similarly-even spatial distribution among populations (Table 7) (Verreault et al., 2005).

Table 7. Mean concentrations of various OC pesticides in adipose tissue of polar bears. (Verreault et al., 2005)

OCP	Alaska	Amundsen Gulf	W. Hudson Bay	Foxe Basin/Gulf of Boothia	Lancaster Sound/Jones Sound	N. Baffin Island	S. Baffin Island	E. Greenland	Svalbard
CBz	116	113	97.5	127	148	191	111	79.1	105
Mirex	8.82	6.87	11.2	1.74	5.59	7.84	4.65	-	10.2
Dieldrin	101	173	152	198	65.2	171	55.4	171	160

Tetrachlorodibenzo-p-dioxin (TCDD) concentrations are highest among populations in the high Arctic, with lower concentrations found in southern populations (Table 8) (Muir et al., 1992).

Table 8. Mean concentrations of TCDD in adipose tissue of polar bears. (Muir et al., 1992)

Location	TCDD (ng/kg)
Beaufort Sea	2
Amundsen Gulf	3
Hadley Bay	11
Melville Island	18
Larsen Sound	23
Barrow Strait	20
Pond Inlet	4
W. Baffin Bay (Clyde River)	5
W. Davis Strait (Broughton Island)	3
Cumberland Sound	< 2
N. Hudson Bay (Coral Harbor)	2
W. Hudson Bay (Rankin Inlet)	2

Within the Canadian Arctic, chlordane and dieldrin concentrations among populations increased from west to east, while concentrations of PCBs were relatively evenly-distributed among populations, with the exceptions of McLure Strait and Eastern Hudson Bay which are possibly receiving local inputs (Table 9) (Muir et al., 1999).

Table 9. Mean concentrations of OCs in adipose tissue of polar bears from the Canadian Arctic. (Muir et al., 1999)

Location	PCB (ng/g lw)	CHL (ng/g lw)	Dieldrin (ng/g lw)
McLure Strait + adjacent Arctic Ocean	20,256	2956	145
Amundsen Gulf and Beaufort Sea to 135˚W	5191	1453	85
Viscount Melville Sound west of 100˚W	8632	1952	96
Queen Maud Gulf and Larsen Sound	4566	2141	147
Barrow Strait and Cornwallis Island	4280	1766	157
Gulf of Boothia	3062	1500	142
Baffin Bay north of 72˚N, Lancaster Sound, Jones Sound, Thule and Ellesmere Island	5985	1987	185
Southern Baffin Bay and Northern Davis Strait	6819	4055	-
Foxe Basin and Hudson Strait west of 72.5˚W	5565	2073	233
Western Hudson Bay (Cape Churchill)	5942	1689	159
Eastern Hudson Bay (Belcher Island)	10,873	4632	335
Davis Strait (below Arctic circle) and Hudson Strait east of 72.5˚W	7049	2074	190

While comparisons with the human tolerable daily intake (TDI) must be made with caution, they can be relatively accurate indicators of biological risk. In the East Greenland population for example, TDI was exceeded for PCBs, dieldrin, chlordane, and HCH (Table 10) (Sonne, 2010).

Table 10. Estimated daily OHC exposure of East Greenland polar bears in relation to TDI. (Sonne, 2010)

Compounds	Concentrations in prey tissue (ng/g ww)	Daily average intake of prey tissue (g)	Daily OHC exposure (µg/kg bw)	TDI (µg/kg bw)
PCB	1186	2740	16.2	0.3
Dieldrin	87	2740	1	0.1
Chlordane	241	2740	3.3	0.05
HCH	74	2740	1	0.3

Data regarding perfluorooctane sulfonate (PFOS) concentrations are limited. However, liver tissue samples from northern Alaskan polar bears have shown mean PFOS levels of 350 ng/g ww. Samples from Hudson Bay bears were approximately 10 times higher in mean PFOS concentration (3.1 µg/g ww). Given the dramatically higher levels, speculation is that there are PFOS sources in close proximity to Hudson Bay. In general, data indicate POP concentrations are highest in the European Arctic and eastern Greenland populations, with lower concentrations found in the Canadian Arctic populations, and the lowest concentrations in the Alaskan Arctic populations (Braune et al., 2005).

Temporal Trends

Temporal trend data for POPs in the Arctic are more limited than spatial data, partially due to the relatively short time periods for which many of the newer POPs have existed. Data on legacy contaminants are more complete. In general, since the 1970s, HCH, CBz, CHL, and DDT concentrations in polar bear fat samples from most locations have decreased significantly. However, PCB concentrations have only slightly decreased and not in all populations. Since the late 1990s, the rate of decrease in concentration of the aforementioned chemicals has greatly slowed. Dieldrin concentrations indicated no consistent temporal trend, with some populations showing increased concentrations, while concentrations decreased among other populations (Table 11) (Verreault et al., 2005).

Table 11. Temporal comparisons between mean concentrations of dieldrin in adipose tissue of polar bears. (Verreault et al., 2005)

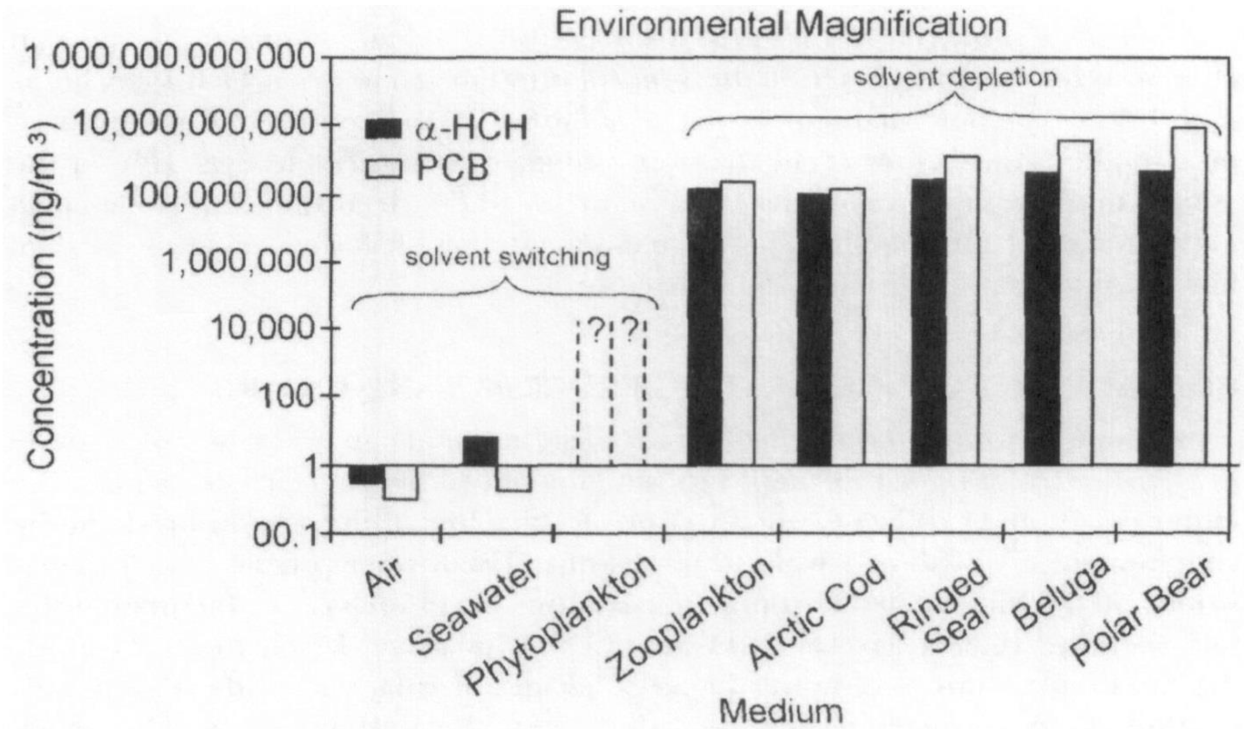
Location	1989 samples	2002 samples	Difference (%)
Alaska	37	99	168
Amundsen Gulf	80	164	105
W. Hudson Bay	107	158	48
Foxe Basin/Gulf of Boothia	216	185	-14
Lancaster Sound/Jones Sound	122	67	-45
N. Baffin Island	138	166	20
S. Baffin Island	174	55	-68
E. Greenland	384	179	-53
Svalbard	189	169	-11

PFCA concentrations among most populations have been consistently increasing for the past decade or more (Sonne, 2010). PBDEs and some other BFRs are a relatively newer class of contaminants and steadily increasing in concentration in most polar bear populations (de Wit et al., 2006). An exception is HBCD, with data indicating decreasing concentrations in many populations, which is likely due to recent changes in regulations and an overall decrease in production (McKinney et al., 2011).

Bioaccumulation/Biomagnification

Since many POPs and their metabolites preferentially bioaccumulate in lipid tissue and aren't easily metabolized, excreted, or otherwise biodegraded, biomagnification readily occurs between trophic levels (Braune et al., 2005). From phytoplankton through apex feeders such as polar bears, each successive trophic level organism bioaccumulates increasing levels of contaminants as a result of solvent depletion, whereby fat is metabolized but contaminants remain and bioaccumulate (Figure 5) (Macdonald et al., 2003).

Figure 5. Environmental magnification in air, water, and the Arctic food web. (Macdonald et al., 2003)



Harsh Arctic conditions require significant caloric intake to meet energetic needs, especially for mammals such as polar bears. The diet of polar bears reflects this necessity because they favorably consume fat and skin, as opposed to muscle tissue, of seals and other marine mammals. Unfortunately, fat and skin are more heavily contaminated with POPs than are other tissues (Sonne, 2010).

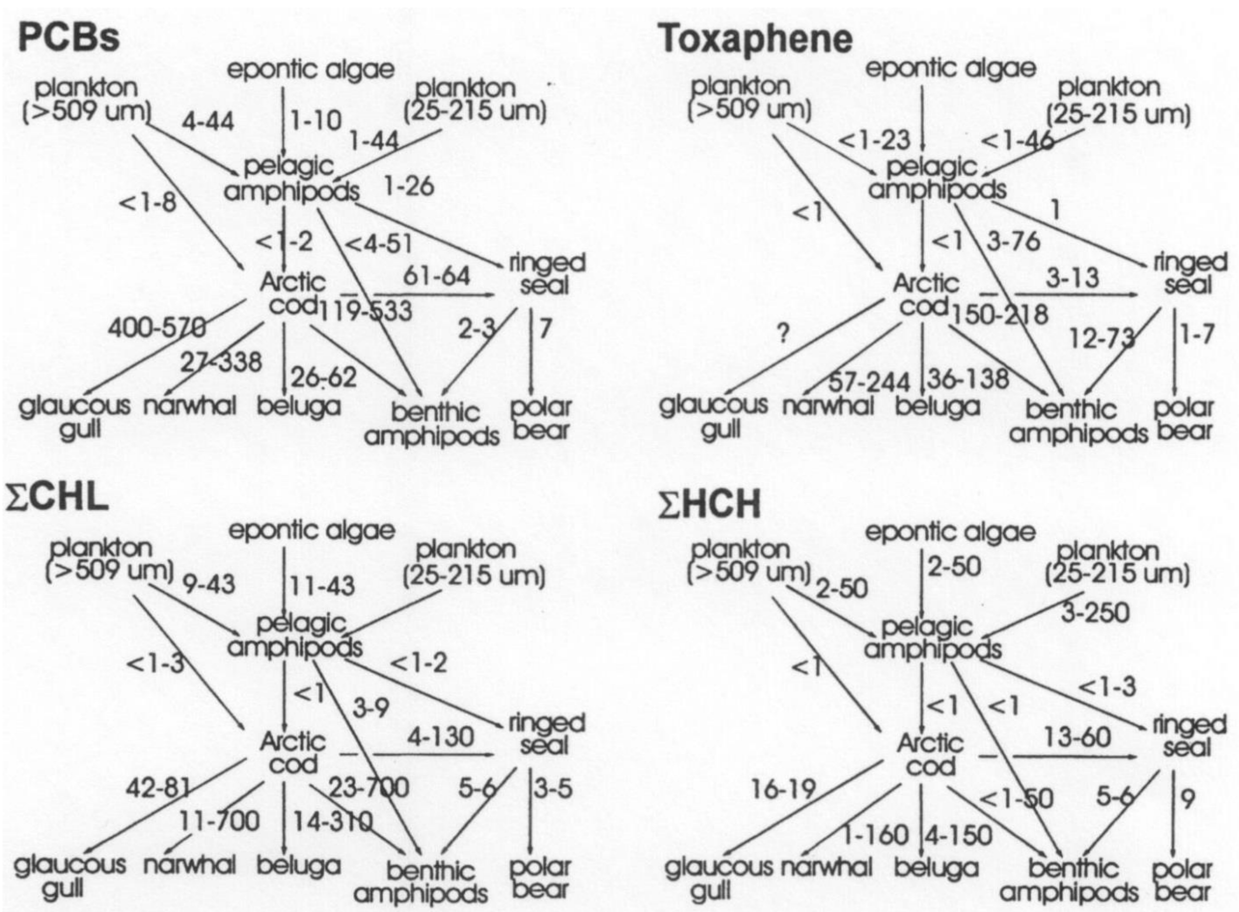
The ratio of a contaminant's concentration in lipid tissue at a given trophic level to that of the next lowest trophic level is referred to as the biomagnification factor (BMF) (Bard, 1999). A BMF greater than 1 indicates biomagnification occurring, while a BMF less than 1 suggests contaminants are being quickly metabolized or otherwise eliminated and not bioaccumulating (Kucklick et al., 2002). BMFs for most POPs range on average from 3-7 per trophic level. Since polar bears are at trophic level 5, the BMF from particulate organic matter in seawater to polar bears can be as high as approximately 17,000 (Table 12) (Muir et al., 1992).

Table 12. Biomagnification factors of various organochlorines. (Muir et al., 1992)

Compound	Water/fish	Fish/seal	Seal/bear
HCH	3.0×10^4	1.5	9.3
CHLOR	5.3×10^7	7.3	6.6
DDT	$\geq 1.4 \times 10^7$	20.2	0.3
PCB	4.8×10^7	8.8	7.4
HCB	9.6×10^6	0.2	15.6

Some of the highest BMFs are those of HCH and PCBs (Figure 6) (Muir et al., 1999).

Figure 6. BMFs of OCs in the Arctic food web. (Muir et al., 1999)



From seals to bears, the calculated BMF of chlordane was 8 in one study. Samples of cod, seal, and bear tissue tested for PCB concentrations, showed obvious biomagnification with respective levels of 0.0037, 0.68, and 4.50 mg/kg (Bard, 1999). Another study calculated a BMF of PCBs of between 2 and 3 from seal to polar bear (Chiu et al., 2000). In contrast, toxaphene does not biomagnify to the same degree, as reflected by relatively low levels in polar bear lipid tissue (Muir et al., 1992).

Variations between populations as to the extent of biomagnified contaminants are influenced by regional differences in diet, food chain structure, and area of home range (Verreault et al., 2005). Bears in off-coast habitats bioaccumulate higher levels of PCBs for instance, than do bears in smaller coastal or near-coastal habitats. This occurs because bears in off-coast habitats have larger territories which require traveling longer distances, thus requiring the consumption of more prey, resulting in more contaminant intake (Willerroider, 2003). For many POPs, concentrations in polar bear tissue increase with age as a result of bioaccumulation (Braune et al., 2005). For example, PCB and DDT concentrations increase with age, particularly in male polar bears (Bard, 1999). PCB concentrations in adult males are typically twice that of females. This is due to PCB maternal transfer from female to cub during parturition and lactation, which lowers the overall PCB body burden of the adult female (Chiu et al., 2000). Since polar bears are relatively long-lived animals, such concentrations frequently reach very high levels (Bard, 1999). Due to their chemical properties, most POPs tend to accumulate in lipid tissue of polar bears, but relatively high concentrations of other POPs, such as PFCAs, may be found in liver tissue, kidney tissue, brain tissue, hair, and blood. Muscle tissue, urine, and feces typically have the lowest POP concentrations in polar bears (Sonne, 2010).

Biological Effects

Vitamins

Some POPs are known to affect vitamin A, E, and D concentrations in polar bear blood and tissue (Letcher et al., 2010). For example, retinol (vitamin A) levels are negatively correlated to PCB concentrations in plasma (Muir et al., 1999). Retinol levels also display negative correlation with hexachlorobenzene (HCB) and HCH concentrations in polar bear blood plasma (Sonne, 2010). Vitamins

A and E are anti-oxidants and critical for growth, development, and health. Vitamin D is associated with bone formation and metabolism (Letcher et al., 2010).

Endocrine System

Many POPs and their metabolites exhibit endocrine disrupting properties and are subsequently classified as endocrine disrupting chemicals (EDCs) (Bechshøft et al., 2012). EDCs may be structurally similar to endogenous hormones, may be able to interact with hormone transport proteins, or may be able to disrupt hormone metabolism, ultimately resulting in chemically-resembling or otherwise negating the effects of endogenous hormones (Jenssen, 2006). Specifically, the synthesis of thyroid and reproductive hormones and cortisol can be seriously affected (Letcher et al., 2010). The endocrine system is critical for numerous hormone-dependent biological processes including reproduction, growth, development, behavior, differentiation, thermoregulation, regulation of metabolism, and immune function (Sonne, 2010). In the case of polar bears, it also aids in adaptation to the extreme Arctic conditions and in responding to environmental stress (Letcher et al., 2010).

Stress stimulates the hypothalamic-pituitary-adrenal (HPA) axis and the neuroendocrine system, which subsequently release glucocorticoids and catecholamines. This is an adaptive response. Some POPs such as PCBs, however, can interfere with the HPA axis, thus compromising adaptation to stress. With respect to PCBs, it has been demonstrated that effects are not strictly a result of total dose, rather the timing of exposure can significantly influence the ultimate effects (Ropstad et al., 2006). The hypothalamus-pituitary-thyroid (HPT) axis and hypothalamus-pituitary-gonadal (HPG) axis are also disrupted by POPs, even with low-level exposure (Bechshøft et al., 2012). In addition to PCBs, other POPs which are classified as EDCs include PCDDs, PCDFs, bisphenol A, PBDEs, tetrabromobisphenol A (TBBPA), phthalates, alkylphenolic compounds, some organochlorine (OC) pesticides, and others (Jenssen, 2006).

Thyroid Hormones

EDCs which chemically and/or structurally resemble thyroid hormones (THs), affect endogenous THs by targeting pathways associated with the HPT axis. The EDCs may interfere with TH production or metabolism, thyroid receptor binding, and cellular uptake and interaction with TH binding proteins (Letcher et al., 2010). THs in polar bears are involved in thermoregulation and the regulation of metabolism during seasonal cycles of adaptive fasting when food availability is reduced. Any disruption of THs, as a result of EDCs, may unfavorably alter the timing of seasonal fasting, which can also impact reproduction.

Studies of bears from the Svalbard and Barents Sea regions showed that PCBs altered five TH variables in females and two TH variables in males, suggesting that females may be more likely to suffer TH-related effects from PCB exposure (Jenssen, 2006). Overall, studies indicate consistent negative correlations between PCBs and thyroid hormone concentrations in polar bear plasma (Skaare et al., 2002). PCBs and their metabolites can impact circulating TH levels, as indicated in a study of Svalbard bears which showed that PCB metabolites had saturated the TH transport capacity in blood. Young polar bears, from those in utero to juveniles, exposed to TH-disrupting POPs may suffer from neurocognitive problems, such as reduced learning ability and altered behavior, as well as disrupted growth, sexual development, and ultimately reduced fertility. Learning to find and hunt prey is critical for long-term survival. These effects not only affect individual health, but population status in general (Bechshøft et al., 2012). Recently, a study discovered thyroid gland lesions among some East Greenland bears. While it has yet to be confirmed, researchers speculate that this a result of EDC exposure and is being caused by either sustained thyroid stimulating hormone (TSH) secretion as a result of HPT axis disruption or suppression of the immune system (Sonne, 2010).

Cortisol

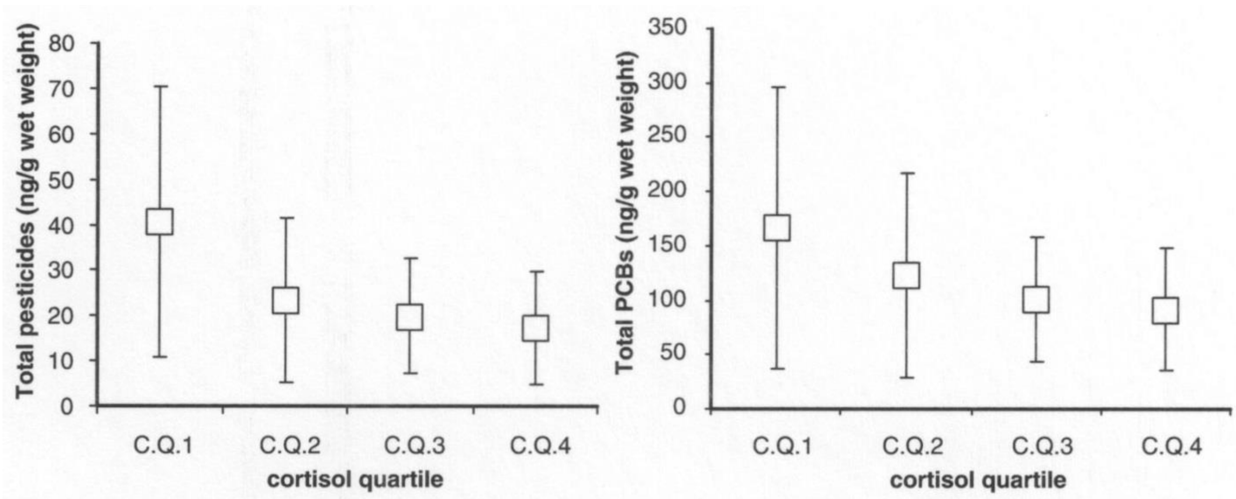
Cortisol is an important hormone influencing growth, development, reproduction, immune function, behavior, metabolism, and stress response (Letcher et al., 2010). Among adult polar bears, females generally have higher cortisol concentrations than males. This difference is not exhibited among other age groups. Plasma cortisol concentrations are positively correlated with both age and axillary girth (Oskam et al., 2004).

Studies of polar bears from Svalbard and the Barents Sea have indicated there is a negative correlation between OC pesticides and plasma cortisol and a positive correlation between PCBs and plasma cortisol. The combined EDC effect on plasma cortisol for these populations, however, showed an overall reduction in circulating plasma cortisol levels (Table 13 and Figure 7) (Oskam et al., 2004).

Table 13. Concentrations of pesticides related to plasma cortisol concentrations in polar bears from Svalbard. (Oskam et al., 2004)

Pesticides	**Cortisol quartile 1: mean 41.1 ng/ml**	**Cortisol quartile 2: mean 101.1 ng/ml**	**Cortisol quartile 3: mean 174.0 ng/ml**	**Cortisol quartile 4: mean 272.7 ng/ml**
HCB	2.9	1.9	2.1	1.8
α-HCH	0.3	0.3	0.2	0.2
β-HCH	1.6	0.9	0.9	0.7
Oxychlordane	34.1	15.6	14.7	14.4
trans-Nonachlor	1.6	1.2	1.2	1.3
p,p'-DDE	0.7	0.5	0.5	0.8

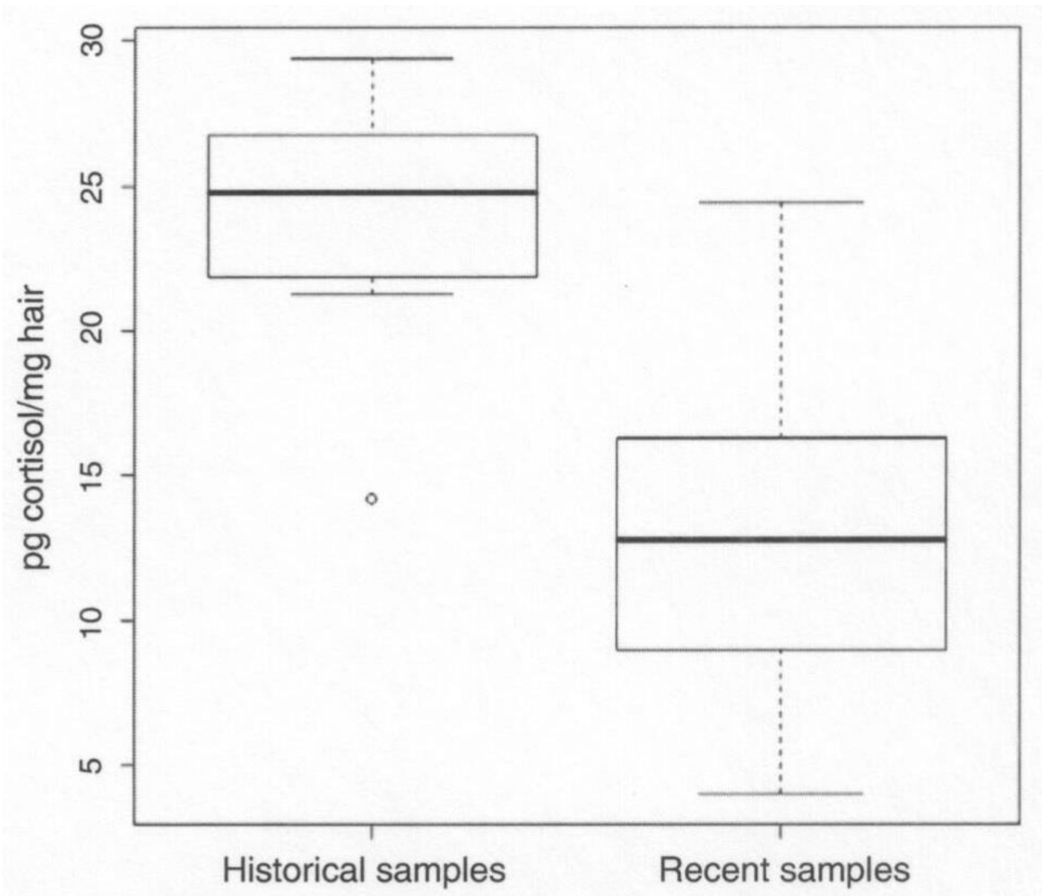
Figure 7. Plasma concentrations of pesticides and PCBs. (Oskam et al., 2004)



These chemicals are somehow modulating the HPA axis, and perhaps the HPT axis (Sonne, 2010). The HPT axis may be affected because of the relationship between TH and cortisol, whereby high cortisol levels can inhibit TH production or sometimes, in contrast, elevated TH levels can promote increased cortisol levels (Bechshøft et al., 2012).

Reduced cortisol levels can inhibit successful adaptation to environmental stress (Jenssen, 2006). A temporal study of cortisol levels among East Greenland bears showed consistently-decreasing levels between 1988 and 2009, with an average annual decrease of 2.7% (Sonne et al., 2012). In fact, using hair samples from archaeological and other preserved polar bear specimens, test results indicate significant decreases in cortisol concentrations between 1892 and 2009. Cortisol concentrations from the period 1892-1927 displayed a mean value of 23.8 pg/mg with a range of 14.19-29.44 pg/mg, while concentrations from the period 1988-2009 displayed a mean value of 12.8 pg/mg with a range of 3.98-24.42 pg/mg (Figure 8) (Bechshøft et al., 2012).

Figure 8. Hair cortisol concentrations in East Greenland polar bears. (Bechshøft et al., 2012)



While cortisol levels can be positively correlated with stress, the aforementioned reductions in cortisol concentrations are not reflective of actual reductions in environmental stress. Rather, the reductions in cortisol are due to EDC exposure, ultimately compromising the ability to adapt to environmental stress as a consequence of reductions in the ability to produce cortisol. These trends in cortisol concentrations correlate with EDC contaminant temporal trends (Bechshøft et al., 2012).

Methylsulphone ($MeSO_2$), which is a metabolite of PCBs and dichlorodiphenyldichloroethylene (DDE), is known to affect cortisol levels and has been found in polar bears. In seals, $MeSO_2$ has been linked to reproductive failure, but this connection has yet to be made for polar bears (Bard, 1999). Furthermore, cortisol is critical for fetal nervous system development and abnormal cortisol levels during this period can result in behavioral abnormalities later in life (Letcher et al., 2010).

Reproductive Hormones

EDCs can affect reproductive hormones by targeting various pathways including steroid binding protein, receptor-binding, or directly impacting ovary or testicular tissues which ultimately impacts the HPG axis (Sonne, 2010). Testosterone is a steroid hormone critical for male sexual development. Studies have determined there is a negative correlation between PCB concentrations and testosterone levels in plasma, indicating that PCB exposure causes reduced testosterone (Skaare et al., 2002). In addition, OC pesticides are negatively correlated with testosterone levels. Surely, the reproductive performance of males with significant levels of PCBs and/or OC pesticides is negatively impacted.

In females, PCB concentrations are positively correlated with progesterone levels in plasma, indicating that PCB exposure increases levels of circulating progesterone (Jenssen, 2006). Progesterone is involved in the regulation of the reproductive cycle by inhibiting the pituitary from releasing follicle-stimulating hormone (FSH) and luteinizing hormone (LH) which are associated with the maturation of follicles (Letcher et al., 2010). As such, increased levels of progesterone may adversely alter the reproductive cycle by preventing normal ovulation from occurring, thus inhibiting successful mating (Jenssen, 2006). In addition to reproduction, testosterone and progesterone also affect sexual behavior (Letcher et al., 2010). Furthermore, exposure to PCBs in utero, and later while nursing, likely alters development of the reproductive system and ultimately adversely affects future reproductive function (Ropstad et al., 2006).

Reproductive System

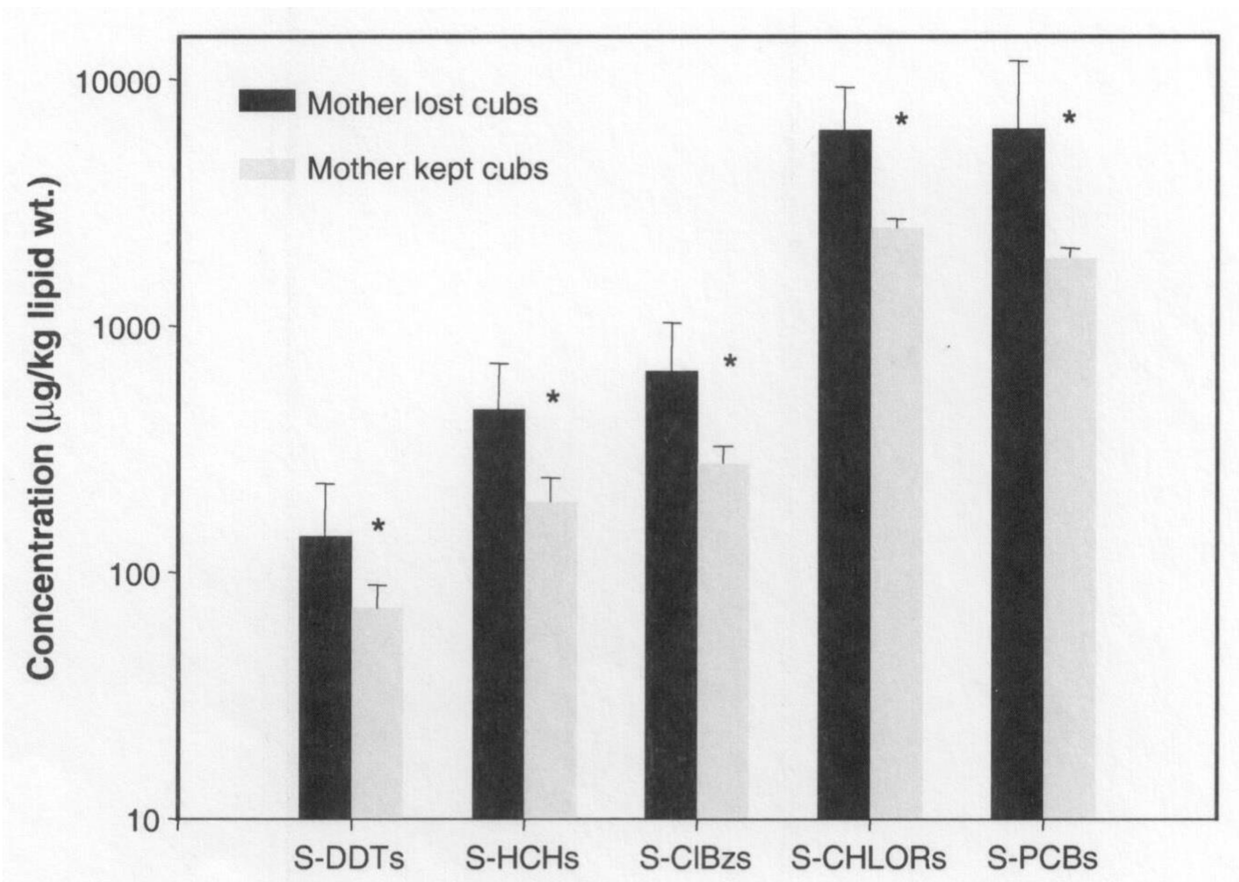
Some POPs are capable of affecting the development and function of sexual organs of both males and females in several ways including reduced organ size, impaired fertility via reduced sperm/egg quality or quantity, and development of lesions. Ultimately, this leads to reductions in birth rates and may impact the status of entire populations (Letcher et al., 2010). For example, PCB exposure is known to damage sperm DNA, which can reduce fertility (Ropstad et al., 2006). In male polar bears, sexual organ size is negatively correlated with adipose tissue concentrations of DDT, dieldrin, CHL, HCH, PCB, HCB, and PBDE. Female sexual organ size is negatively correlated with PCB, CHL, PBDE, and HCB. Reduced organ size has been associated with OC concentrations as low as 13 ng/g lw and PBDE concentrations as low as 22 ng/g lw (Sonne, 2010). In one study, hermaphroditism was documented in at least four female bears in Svalbard, and speculation is that it may be a result of high POP exposure during the fetal stage or early development, possibly causing elevated levels of circulating testosterone (Letcher et al., 2010). Other data indicate that approximately 2%, or 100 bears, of the Svalbard population are hermaphrodites, with conjecture that PCB exposure is the cause (Willerroider, 2003).

Maternal Transfer

Perhaps one of the most significant biological impacts of POPs on polar bears is that which occurs during the critical fetal and early developmental stages, as facilitated by the maternal transfer of contaminants to offspring. Such contaminants can adversely affect the development of the endocrine, reproductive, immune, nervous, and other body systems, creating life-long health problems and threatening individual survival and the population in general (Oskam et al., 2004). Females fast for approximately four months during the end of the gestation and early lactation periods. Placental and lactational transfer is enhanced during times of fasting when loss of fat increases concentrations of circulating levels of contaminants (Bard, 1999). One study determined that Hudson Bay females lost approximately 24-29% of their CHL and PCB body burden, on average, during the fasting period. Most of these contaminants were probably transferred to offspring via placental and later lactational transfer (Fisk et al., 2005).

Milk is rich in lipids and consequently hydrophobic contaminants. Developing fetuses have relatively low quantities of fat reserves, which reduces potential storage of contaminants and results in sustained circulation of bioavailable contaminants in the blood (Bytingsvik et al., 2012). Adult females typically have lower PCB concentrations than males and this is likely due to a direct transfer to offspring during parturition and lactation (Chiu et al., 2000). This loss of contaminants through maternal transfer is further indicated by a study comparing OC concentrations in spring and fall of mothers with cubs and mothers who had lost their cubs during the year. OC concentrations in the fall of mothers who had lost their cubs were significantly higher than in mothers who had been nursing throughout the year, suggesting the reduced concentrations in the latter group were due to losses via lactation (Figure 9) (Fisk et al., 2005).

Figure 9. Major OCs in milk of females with cubs after emerging in March from dens. (Fisk et al., 2005)



With respect to PCB concentrations in the same study, levels in females which had lost their cubs (5780 ng/g lw) were approximately 3 times higher than those of females who kept their cubs (1830 ng/g lw) (Fisk et al., 2005).

Elevated levels of CBz, CHL, and HCH have been found in the milk of females from populations with relatively high levels of these contaminants, further suggesting the extent of potential maternal transfer of POPs (McKinney et al., 2011). One study determined that up to 70% of total OC body burden is maternally transferred to offspring during lactation. Additionally, first-born cubs receive the highest concentrations of POPs (Sonne, 2010). In general, cubs typically have twice the body burden of POPs as their mothers (Bard, 1999). Reduced cub survival has been documented in the Svalbard population, with high PCB exposure being the speculated cause (Skaare et al., 2002). In fact, data from a 2008 Svalbard study determined that mean PCB concentrations in cubs averaged 2.7 times that of their mothers (Bytingsvik et al., 2012). CHL and PCB concentrations are usually higher among subadults than adults, and this is likely a result of remnant concentrations ingested during lactation earlier in life (Verreault et al., 2005).

Immune System

Interconnected relationships exist between the immune system and internal organs, body tissues, and hormones such as cortisol. The proper functioning of this interconnected system is necessary to handle environmental stress such as that caused by exposure to contaminants and climate change. Unfortunately, many contaminants directly and significantly impair proper immune system response which may threaten individual survival and overall population size as a result of increased susceptibility to disease (Sonne, 2010).

The specific (acquired) arm of the immune system consists of humoral and cell-mediated immune responses (Fisk et al., 2005). Antibodies such as immunoglobulin G (IgG) are a product of humoral immune response and are critical for protecting against infections (Skaare et al., 2002). Studies of Svalbard bears have determined a negative correlation exists between PCBs and IgG suggesting that PCB exposure reduces antibody production and subsequently reduces immune system function (Oskam et al., 2004). The Svalbard bear population, on average, has significantly higher PCB concentrations than those found in Canadian populations. A study was conducted to evaluate and compare antibody reactions to immunizations in the Svalbard and Canadian bears. The Canadian bears, with lower respective concentrations of PCBs, produced notably more antibodies following immunization than did the Svalbard bears who have high PCB concentrations, further indicating the negative correlation between PCBs and antibodies and the increased risk of infections for bears with high levels of PCBs (Skaare et al., 2002). Additionally, cell-mediated immune response is negatively affected by PCBs and OC pesticides as a result of a reduction in the production of cytotoxic T-cells which attack viruses and tumors (Fisk et al., 2005). Natural killer cells (NKC) are a component of the non-specific (innate) immune response and also indicated to be suppressed by the presence of PCBs. OC concentrations as low as 1400 ng/g lw are known to greatly reduce specific and non-specific immune response in polar bears (Sonne, 2010).

Internal Organs

The liver is involved in detoxification, protein synthesis, the production of biochemicals for digestion, and metabolism. Water and electrolyte conservation and clearance of metabolic waste products are two of the primary kidney functions. Damage to the functions of liver or kidneys is generally irreversible and threatens individual health and survival which may ultimately impact population status (Sonne, 2010).

Liver and kidney damage in polar bears has been attributed to POP exposure, likely as a combined result of direct toxicity and immunosuppressive effects subsequently increasing rates of infection. Specifically, HCB and HCH appear to increase the frequency of both renal and hepatic lesions, while PCB and PFOS have been linked to hepatic lesions (Sonne et al., 2012). In a study of the East Greenland population, HCH and HCB adipose tissue concentrations as low as 50 ng/g lw and PCB concentrations as low as 6000 ng/g lw have been positively correlated to the occurrence of liver lesions. The same study determined positive correlations between the presence of renal lesions and PBDE adipose tissue concentrations as low as 50 ng/g lw as well as OC pesticide concentrations as low as 150 ng/g lw. Furthermore, temporal data indicate increasing hepatic PFCA concentrations of up to 27% per year in the same population, which may suggest another contaminant-associated stimulus for the increasing prevalence of hepatic lesions (Sonne, 2010).

Skeletal System

Renal lesions, a possible effect of POP exposure, can increase the rate of loss of calcium, phosphorus, magnesium, and various proteins, which are the major constituents of bone. As a result, bone mineral density (BMD) will be reduced over time. Another mechanism whereby POP exposure may lead to reduced BMD is by POP-induced increases in cortisol concentration which decreases the amount of calcium taken up by the small intestines, leading to bone loss or osteoporosis, ultimately increasing susceptibility to bone fractures which may threaten survival. Prenatal exposure to some POPs is known to impair the development of teeth and cause bone asymmetry in skulls (Sonne, 2010).

In controlled, laboratory studies of various species, POPs such as PCBs and DDT have been linked to detrimental effects on the skeletal system such as reduced BMD and periodontitis. In a study of East Greenland bears, concentrations of PCBs, DDT, PBDE, HCB, dieldrin, and chlordane were all negatively correlated with BMD (Tenenbaum, 2004). Chlordane and PCB concentrations as low as 1500 ng/g lw and dieldrin, HCB, and HCH concentrations as low as 50 ng/g lw have been attributed to a decrease in polar bear skull BMD (Sonne, 2010). Another study involving testing of historical samples has determined that the BMD and overall size of male East Greenland polar bear skulls has steadily decreased over the past 120 years (Sonne et al., 2012). Not surprisingly, this period corresponds with the introduction and rapid accumulation of anthropogenic contaminants, such as POPs, in East Greenland biota. A more recent temporal investigation determined that the size decreased and asymmetry increased of East Greenland polar bear skulls during the period 1992-2002, with POP exposure speculated to be the major contributing factor (Sonne, 2010).

Neurological and Behavior Effects

Behavior, learning, and cognitive abilities may be affected by exposure to some POPs. Data are scarce with respect to the effects of POPs on polar bear neurology and behavior. Nevertheless, neurodevelopmental and behavioral disorders resulting from TH disruption from EDC exposure have been documented in humans and laboratory animals (Letcher et al., 2010).

EDCs which affect TH in polar bears, particularly during developmental life stages, may disrupt learning such that successful hunting later in life is reduced (Jenssen, 2006). East Greenland and Svalbard bears, with the highest levels of PCBs and correspondingly low TH levels, are therefore expected to suffer from neurodevelopmental and behavioral disorders (Letcher et al., 2010). In addition, POPs that affect behavior and cognitive abilities may create difficulties for bears adapting to climate change-induced alteration of ice-coverage which will affect hunting and migration (Jenssen, 2006).

Biotransformation and Enzyme Systems

Biotransformation is a process which occurs in organisms whereby chemical substances, such as POPs, are metabolized or otherwise degraded, resulting in metabolites which may be more or less toxic than their parent compounds (Letcher et al., 2010). Therefore, a highly-functional metabolic system does not necessarily indicate a reduction in health impacts associated with POP exposure because even if concentrations of parent compounds decrease over time, toxic metabolites may persistently bioaccumulate (Jenssen, 2006). Furthermore, it is important to remember, the levels of chemicals which are effectively metabolized in an organism may not accurately reflect the levels of those chemicals in the environment.

One method to determine an organism’s ability to metabolize a chemical is by comparing the levels of metabolic products with the level of the parent compound (Bard, 1999). Many metabolites known to be present in polar bears are in fact as or more persistent, bioaccumulative, toxic, and biologically active as their parent compounds, including examples such as heptachlor epoxide, oxychlordane, OH-PCB, $MeSO_2$-PCB, and OH-PBDE (Letcher et al., 2010). Similar to their parent compounds, many POP metabolites structurally resemble endogenous compounds and are capable of disrupting growth, development, vitamin homeostasis, hormone production, the reproductive system, and other body systems (Bytingsvik et al., 2012).

Biotransformation occurs primarily in the liver and is mediated by enzymes such as the cytochrome P450 (CYP) group of enzymes (Letcher et al., 2010). In polar bears, CYP enzymes are known to biotransform parent compounds such as OCs, PBDEs, and PFCAs, among others, into highly toxic metabolites (Sonne, 2010). Chemicals or physiological states which adversely affect the functioning of enzyme systems will reduce metabolic capability (Letcher et al., 2010). CYP activity is used as a biomarker to gauge the ecological significance and biological susceptibility of a given contaminant based on the physiological response of an organism to exposure (Bard, 1999).

Many metabolites are known EDCs, including OH-PCB, $MeSO_2$-PCB, 3-$MeSO_2$-*p,p'*-DDE, 4-OH-heptachlorostyrene, pentachlorophenol, and OH-PBDE, all of which have been found in fat, liver, and brain tissue and blood of Canadian and East Greenland bears (Letcher et al., 2010). In some cases, metabolites target and accumulate in different tissue than do their parent compounds. For example, OH-PCBs tend to accumulate in blood, while PCBs are associated with lipids. In fact, this is the case for bears in Svalbard, East Greenland, and the Canadian Arctic, where studies have indicated OH-PCB levels in blood are significantly higher than those of PCBs (Bytingsvik et al., 2012). When evaluating the contaminant load in and subsequent health impacts on polar bears, it's important for researchers to test for metabolites in addition to the typical parent compounds so as to avoid errors in interpretation of toxicity and effects (Sonne, 2010).

CURRENT-USE/NEW CHEMICALS

As the production and use of certain chemicals is banned, phased out, or otherwise voluntarily reduced or eliminated, new chemical formulations are released to the market. Frequently, these new compounds are initially claimed to be more environmentally-friendly or even entirely risk-free. However, as is usually the case, data inevitably arise showing these claims are false and unintended environmental damage has taken place or is continuing to occur as a result of intentional releases as well as accidental leaks (McKinney et al., 2011).

With respect to some recent- or current-use chemicals though, environmental data are very limited, especially spatial and temporal trends, species differences, sources, pathways, and food web dynamics (Braune et al., 2005). What has been established is that many of these chemicals are being transported to and accumulating in the Arctic ecosystem. Short-chained chlorinated paraffins (SCCPs), PCNs, PFOS, and the previously-discussed group of BFRs are a few examples. The latter three formulations are found in many consumer products. Some formulations classified as BFRs include polybrominated biphenyls (PBBs), pentabromotoluene (PBT), pentabromoethylbenzene (PBEB), hexabromobenzene (HBB), 1,2-*bis*(2,4,6-tribromophenoxy)ethane (BTBPE), and decabromodiphenyl ethane (DBDPE) (McKinney et al., 2011).

Data indicate the presence of several current-use pesticides in seawater, ice, and the atmosphere in the regions of the Bering and Chukchi Seas. These pesticides include chlorothalonil, trifluralin, atrazine, chlorpyrifos, and endosulfan (Bard, 1999). Atrazine inhibits photosynthesis and therefore can impact primary production. Two additional pesticides which have been detected in the Arctic are methoxychlor and pentachlorophenol. Methoxychlor has been demonstrated to have estrogenic activity. Pesticide use in agriculture continues to increase globally, thus increased accumulation of current-use pesticides in the Arctic is expected (Muir et al., 1999). The solution to chemical contamination of the Arctic is likely a reduction in production and use, rather than the creation of new chemical products.

METALS

Heavy metals such as lead, cadmium, mercury, arsenic, and others are of significant biological concern to Arctic biota. Elevated levels of some metals cause numerous negative health impacts among mammals and other organisms such as reproductive problems, damage to liver and kidney tissues, neurological effects, and a variety of other consequences. Much of the Arctic anthropogenic metal contamination is a result of wet scavenging and dry deposition as a consequence of aerosols originating from industrial sources in Europe and Asia (Bard, 1999).

Lead contamination is a global problem. Primary sources of aerosols containing lead include smelters, gasoline exhausts, and various industrial activities. Industrial liquid wastes and sewage outfalls contribute lead directly to bodies of water in which they are released (Muir et al., 1992). Mercury contamination has been a concern for many decades. However, after two major incidents of methyl mercury poisoning in Japan in the mid-20th century, greater attention was directed to the environmental and health consequences of anthropogenic mercury pollution (Eaton and Farant, 1982). Common sources of mercury contamination include combustion of coal and municipal garbage. Because of its high volatility and relatively long half-life, atmospheric mercury transport is of significant magnitude. In North America, for example, much of the anthropogenic mercury north of Hudson Bay originated from coal combustion in the mid-west (Bard, 1999). Other potential sources of metals include mine-tailing discharge, off-shore oil production facilities, and municipal waste disposal sites.

Determining the extent of anthropogenic heavy metal contribution to the environment can be more challenging than that of POP contribution, however. While both global and local sources of contaminants may contribute heavy metals to the sediment in a given area, metals are naturally-occurring and concentrations are a result of local geology, particle size, organic matter content, and other factors (Muir et al., 1992). To illustrate this variability, pre-industrial mercury concentrations in Arctic lakes range from 0.7 to 54.35 $\mu g/m^2/yr$ (Bard, 1999). As a result, it is often difficult to distinguish between “normal” background levels and elevated levels as a result of anthropogenic addition (Muir et al., 1992).

Metals can adsorb to particles of sediment. As organisms ingest sediment, these metals bioaccumulate in tissues, ultimately biomagnifying up the food chain (Bard, 1999). With respect to metal contamination in biota, there are practically no areas on earth, save parts of Antarctica perhaps, which have not received some level of anthropogenic metal contaminants. Because of this, it is impossible to gauge the extent of metal contamination by comparing individuals of one species from a contaminated area to individuals of the same species from an uncontaminated area. Fortunately, scientists in some cases have been able to compare metal concentrations in living animals to concentrations from prehistoric samples. This comparison aids in approximating the extent of anthropogenic metal contamination in biota since prehistoric times (Muir et al., 1992). Comparisons can also be made between tissues and hair of living animals to those of dated museum specimens in order to compare historic levels. Broad comparisons between marine mammals and domestic and laboratory animals have indicated elevated cadmium and mercury levels among marine mammals (Woshner et al., 2001). Similar to background metal levels in the environment, tissues of biota also contain some levels of metals regardless of anthropogenic contribution.

In mammals, there are some relatively common correlations involving metals, tissues, and age. For example, there are generally positive correlations between the concentrations of mercury and selenium in liver, age or length and cadmium concentration in kidney and liver, age and mercury concentration in liver and kidney, age and selenium concentration in liver, age and zinc concentrations in kidney, and concentrations of zinc and copper in kidney or liver (Muir et al., 1992). Iron is one of the only elements that routinely varies in concentration based on sex of the animal, as it is usually lower in livers of females (Muir et al., 1999). These associations must be accounted for when determining metal contribution from anthropogenic sources. For instance, organic mercury in polar bear kidney is < 6% of total mercury, while organic mercury in adult seal and whale kidney averages 10-20% of total mercury (Muir et al., 1992).

Cadmium, mercury, and selenium concentrations in marine mammal tissue are frequently found to exceed levels which are known to be toxic in domestic animals (Table 14) (Fisk et al., 2005).

Table 14. General threshold levels of biological effects for mammals. (Fisk et al., 2005)

Metal	Tissue	Concentration (μg/g ww)	Effect
Cadmium	Kidney	> 50	Potential renal dysfunction
Mercury	Liver	> 25	Liver damage
Selenium	Liver	> 7	Hepatic lesions

In fact, Woshner et al. (2001) found mean selenium and mercury concentrations in liver and kidney tissue of ringed seals and polar bears to exceed levels associated with toxicity in domestic animals. To some degree, hair samples are another means to analyze metal contamination. Generally, if concentrations of metals in hair are elevated above background levels, then concentrations in tissues such as liver can be expected to be elevated as well. However, it is important to note that concentrations in tissue are cumulative, while concentrations in hair simply reflect metal concentrations in blood at the time of hair growth (Eaton and Farant, 1982).

Spatial Trends

Spatial trend data are somewhat limited for heavy metal contamination in the Arctic, as compared to that of POP contamination. Mercury concentrations in polar bears spanning the entire Arctic range from 13.4 to 99.8 μg/g ww, with the highest levels found generally in the northern and western Arctic (Table 15) (Muir et al., 1992).

Table 15. Mean mercury concentrations in polar bears from various locations. (Muir et al., 1992)

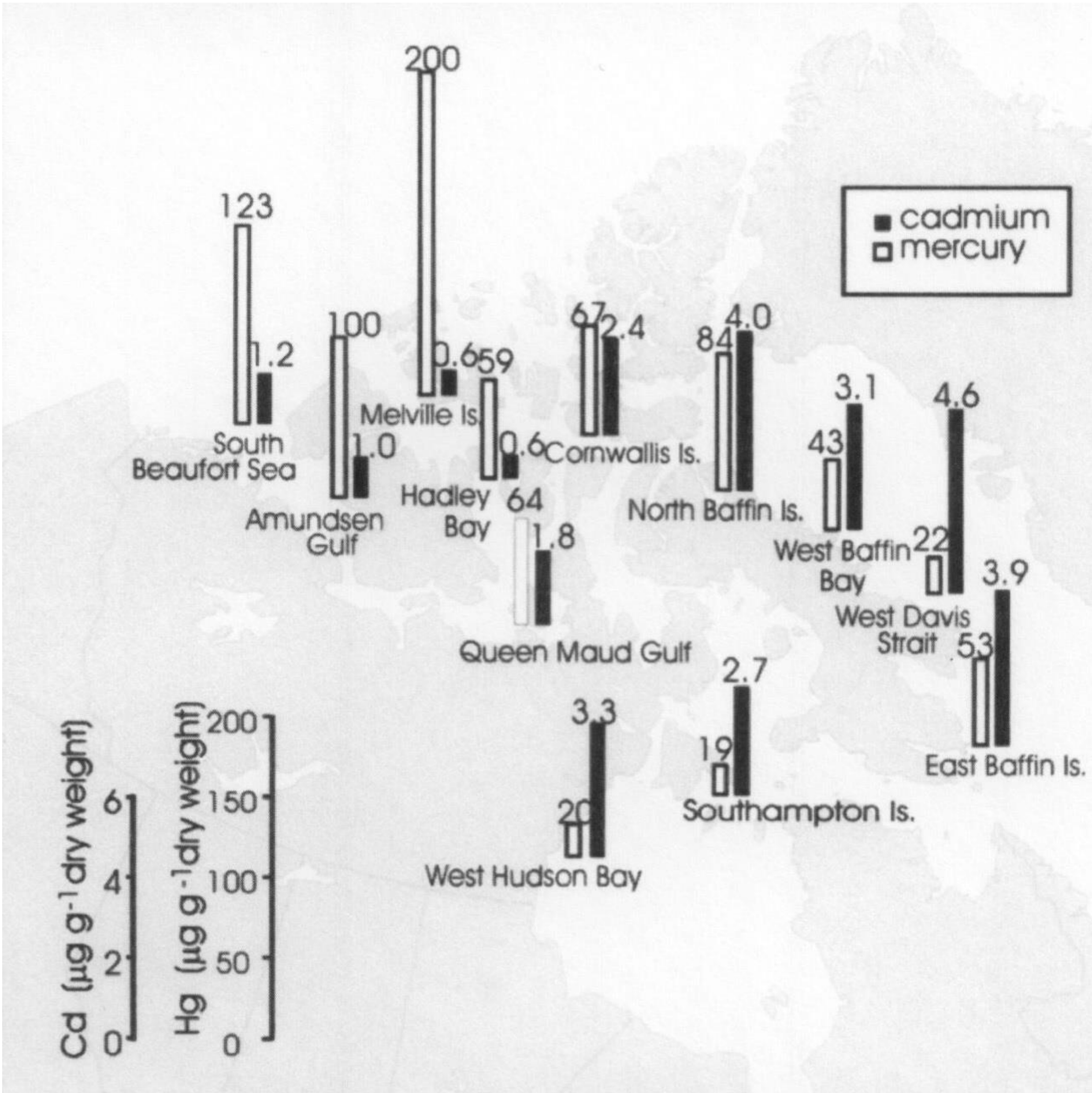
Location	Mercury (µg/g ww)
Jones Sound	40.1
Spence Bay region	36.7
Melville Island	99.8
Hadley Bay (Victoria Island)	44.9
Amundsen Gulf	64.0
S. Beaufort Sea	45.4
W. Greenland	13.4

On average, mercury concentrations of western Artic bears are seven times higher than those of eastern Arctic bears (Muir et al., 1999). Concentrations predictably decrease following a southern and eastern gradient.

These differences in mercury concentration may be due to differences in the diets of the respective polar bear populations. Bears in western and northern regions of the Arctic tend to consume more bearded seal, as opposed to bears in the eastern Arctic which primarily consume ringed seal. Likely as a result of bearded seals significant consumption of benthic invertebrates in contrast to that of ringed seal, bearded seal typically have higher levels of mercury in liver tissue (Eaton and Farant, 1982). It is probable that the higher mercury concentrations in sediment in the western Arctic may be primarily due to natural, geologic origin. M'Clure Strait presents an anomaly among mercury concentration data in the Canadian Arctic, as concentrations in livers of polar bears in this region are vastly higher than those found in nearby regions. There are no lower food web contaminant data available to determine the exact causes, but speculation is that bearded seals may be more locally abundant in this area relative to the rest of the region.

Similar to mercury, selenium concentrations follow the same geographic pattern. Selenium concentrations in western Arctic bears average ten times the concentrations found in polar bears in the eastern Arctic. In contrast to mercury and selenium, cadmium concentrations in bears from the western Arctic are approximately three times lower than in eastern Arctic bears (Figure 10) (Muir et al., 1999).

Figure 10. Cadmium and mercury in polar bear liver from the Canadian Arctic. (Muir et al., 1999)



These differences in cadmium concentrations may also be a result of geological differences, with the eastern Arctic having higher levels of cadmium in sediments. Another possibility is the diets of seals, regardless of species, may vary enough between regions to account for some of the differences in metal concentrations in polar bears, since these metals are bioaccumulative (Muir et al., 1999).

Temporal Trends

Temporal trend data of metal concentrations in the Arctic are also somewhat limited as compared to that of POP data. Nevertheless, the general consensus holds that most heavy metals have increased in the Arctic since prehistoric times. Lead is one of the most anthropogenically elevated metals in the northern hemisphere. With respect to lead levels, estimates indicate prehistoric oceanic surface water levels were approximately 0.6 ng/l. Current levels in the North Pacific, for example, are ten times greater than prehistoric levels, and in the Arctic the levels are even greater. Lead concentrations in the Greenland ice sheet are approximately 300 times that of prehistoric levels.

Mining operations are notorious for contributing to heavy metals contamination. Temporal trend data show that in only four years, from the commencement of operations at the Nanisivik mine in 1975 to 1979, zinc, lead, and cadmium concentrations drastically increased in sediment in Strathcona Sound and the nearby Beaufort Sea (Muir et al., 1992). Evidence indicates current mercury levels in the Arctic are certainly higher than during pre-industrial times (Braune et al., 2005). Sediment studies of northern lakes indicate minimal mercury increase from 1500 to 1750 AD, and these levels are accepted as natural, background levels. From approximately 1750 to the present, sediment samples indicate rapid increases in mercury concentration, which coincides with the onset of the Industrial Revolution. Therefore, these elevated levels are indicative of anthropogenic sources. Sediment samples from an Arctic lake in Imitavik show current mercury levels are six times higher than in the mid-18th century (Bard, 1999).

However, temporal trends over the past two to three decades, particularly in the Canadian Arctic, are highly variable and inconclusive. Some populations experienced increasing mercury concentrations during this period of time, while concentrations among other populations were seemingly unchanged. Therefore, at least for the Canadian Arctic, the limited current data preclude an accurate determination of recent mercury origin, whether anthropogenic or natural. Pre-industrial polar bear hair samples from archaeological sites have been compared to hair samples from modern bears, with the latter showing mercury concentrations several times greater (Braune et al., 2005). Atmospheric mercury depletion events (MDEs) have garnered significant study as to their potential contribution to contamination of those populations with currently increasing mercury concentrations. However, many scientists agree that methyl-mercury is a more likely source for marine mammal populations exhibiting recent increases in mercury concentration (Macdonald, 2005).

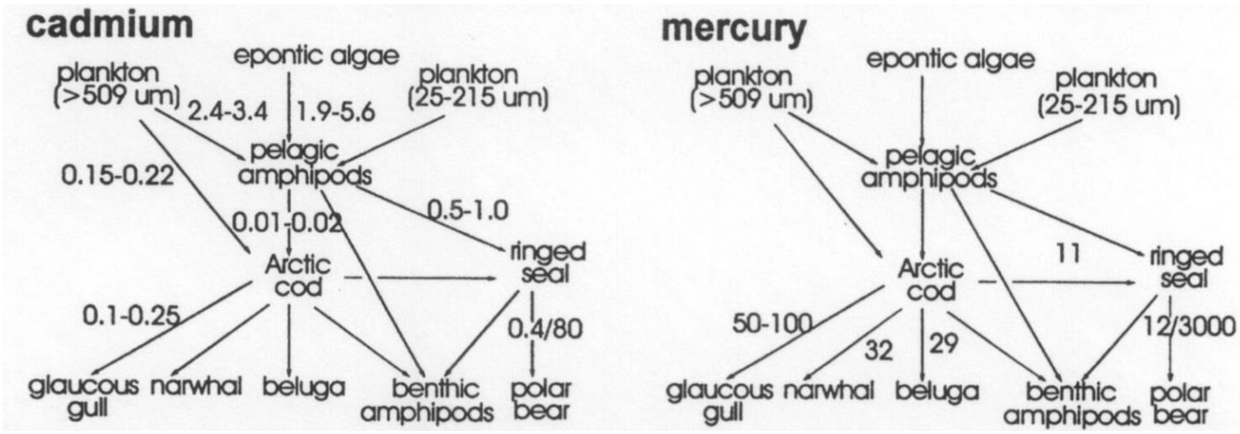
Bioaccumulation/Biomagnification

As with POPs, metals are capable of bioaccumulating and biomagnifying up the food chain. In the case of mercury, benthic invertebrates ingest mercury-contaminated sediment (Eaton and Farant, 1982). Uptake of metals varies depending on the elemental species, which may be modified by changes in water hardness, salinity, redox conditions, pH, and temperature (Braune et al., 2005). Fish and seals feed on these invertebrates, and seals also feed on contaminated fish, thus incorporating mercury in their respective body tissues. Ultimately, polar bears receive large levels of mercury contamination as a result of feeding on the aforementioned fish and seals (Eaton and Farant, 1982).

While the Arctic has less mercury contamination in relation to many other regions of the earth, mercury somehow manages to significantly bioaccumulate in Arctic biota. This is a result of the low primary productivity in the Arctic. Assuming a hypothetically static level of mercury (or any other metal) in a given body of water, as the concentration of phytoplankton decreases, the mercury content of said phytoplankton will concurrently increase. Therefore, this explains how mercury is so efficiently bioaccumulated and biomagnified up the food chain in the Arctic even though actual mercury levels in sediment and water are lower than that of other oceans (Bard, 1999). While mercury contamination is of significant concern, other potentially toxic metals are known to bioaccumulate and incorporate in various polar bear body tissues.

BMFs between bears and seals are significant and evidenced by notably increased metal concentrations in polar bear tissue as compared to that of seals (Figure 11) (Muir et al., 1999).

Figure 11. BMFs of cadmium and mercury in the Arctic food web. (Muir et al., 1999)



Metals vary in which tissue they primarily accumulate (Woshner et al., 2001). Furthermore, different organisms may accumulate the same metal in differing tissues. In part, this is due to the effects that different metabolic rates have on elemental transfer across biological membranes. For example, since fish and polar bears have drastically different metabolic rates, metals will differ in which tissues they primarily bioaccumulate (Braune et al., 2005).

In general, among mammals, copper (Cu) bioaccumulates most in liver and muscle, manganese (Mn) in liver, molybdenum (Mo) in liver, silver (Ag) in liver, arsenic (As) in liver, cadmium (Cd) in kidney, cobalt (Co) in kidney, lead (Pb) in kidney, magnesium (Mg) in muscle, selenium (Se) in kidney and muscle, zinc (Zn) in liver and muscle, and mercury (Hg) in liver. Metal bioaccumulation in polar bears follows a similar pattern, however, for unknown reasons, mercury bioaccumulates to a greater degree in kidney, as opposed to liver in polar bears (Table 16) (Woshner et al., 2001).

Table 16. Concentrations (µg/g ww) of various elements in tissues of polar bears. (Woshner et al., 2001)

Tissue	As	Cd	Cu	Pb	Mg	Mn	Se	Ag	Zn	Hg
Liver	0.09	0.47	30.0	0.08	371.0	5.13	9.33	0.17	78.64	14.22
Kidney	0.02	8.69	3.39	0.29	247.0	1.80	12.99	0.01	39.60	NA
Muscle	ND	0.01	2.97	0.02	495.3	0.28	0.54	ND	64.08	0.09
Blubber	0.07	ND	0.27	ND	27.94	0.05	0.04	0.01	1.96	NA

NA – Not analyzed ND – Not detected

To illustrate, seal to bear BMFs for mercury are 1.99 in liver and 11.6 in kidney (Woshner et al., 2001). It is important to highlight, these BMFs are based on the ratios of concentrations in the respective organs of each species, and therefore not reflective of complete body-burden BMFs. In some eastern Canadian populations, non-tissue specific BMFs from seal to bear have been estimated at approximately 3000 for mercury and 80 for cadmium (Muir et al., 1999). In polar bears, mercury also bioaccumulates to a significant degree in hair and blood, while muscle, urine, feces, and brain tissue generally exhibit minimal bioaccumulation (Sonne, 2010). Bioaccumulation in body tissues is further exacerbated by the relatively long half-lives of some metals. For example, in human kidneys, the half-life of cadmium is known to be over 10 years. Such data are not available for polar bears, but it is likely a similar value (Muir et al., 1992).

Biological Effects

Polar bear fetuses and cubs may be exposed to metal contamination through maternal transfer while in the womb or via nursing. Depending on the metals and levels of exposure, permanent damage to bodily systems can occur during these critical stages of growth and development. For example, prenatal mercury exposure can disrupt nervous system development, which then manifests in altered behavioral patterns.

Mercury can cause permanent damage to both kidney and liver function. For terrestrial mammals, the general toxic threshold level of mercury is 30 µg/g ww. Among Arctic species, polar bears consistently have the highest kidney mercury concentrations. Kidneys serve multiple physiological functions such as clearing metabolic waste products and conserving water and electrolytes. Decreased kidney function can ultimately lead to renal failure and death. Sonne (2010) found that 9 of 57 (16%) sampled individuals had renal tissue exceeding 30 µg/g ww mercury concentration. The mean concentration among the samples was 14.3 µg/g ww, with a range from 1 to 50 µg/g ww. In polar bears, kidney mercury concentrations as low as 14,300 ng/g ww are known to cause kidney lesions, and liver mercury concentrations as low as 10,500 ng/g ww are known to cause liver lesions.

In mammals, the major excitatory neurotransmitter receptor in the brain is glutamate-NMDA. Sonne (2010) found a marked negative correlation between glutamate-NMDA receptor levels and both total mercury and methylmercury. Since glutamate stimulates gonadotropin-releasing hormone (GnRH) and luteinizing hormone (LH) release, any disruption of glutamate-NMDA receptor levels as a result of mercury exposure may result in the alteration of sex hormone concentrations, delayed onset of puberty, and changes in reproductive behaviors. Changes in glutamate levels may also disrupt innate mechanisms in polar bears to appropriately respond to changes in the environment. In turn, this could create energetic stress and potentially skew migration patterns.

In his analyses of mercury concentration in brains of polar bears, Sonne (2010) found the highest concentrations in the pituitary glands. If pituitary function is compromised, numerous body systems may be negatively impacted, since the pituitary gland regulates the release of all hormones via the neuro-endocrine hypothalamic-target organ feed-back loops (Sonne, 2010).

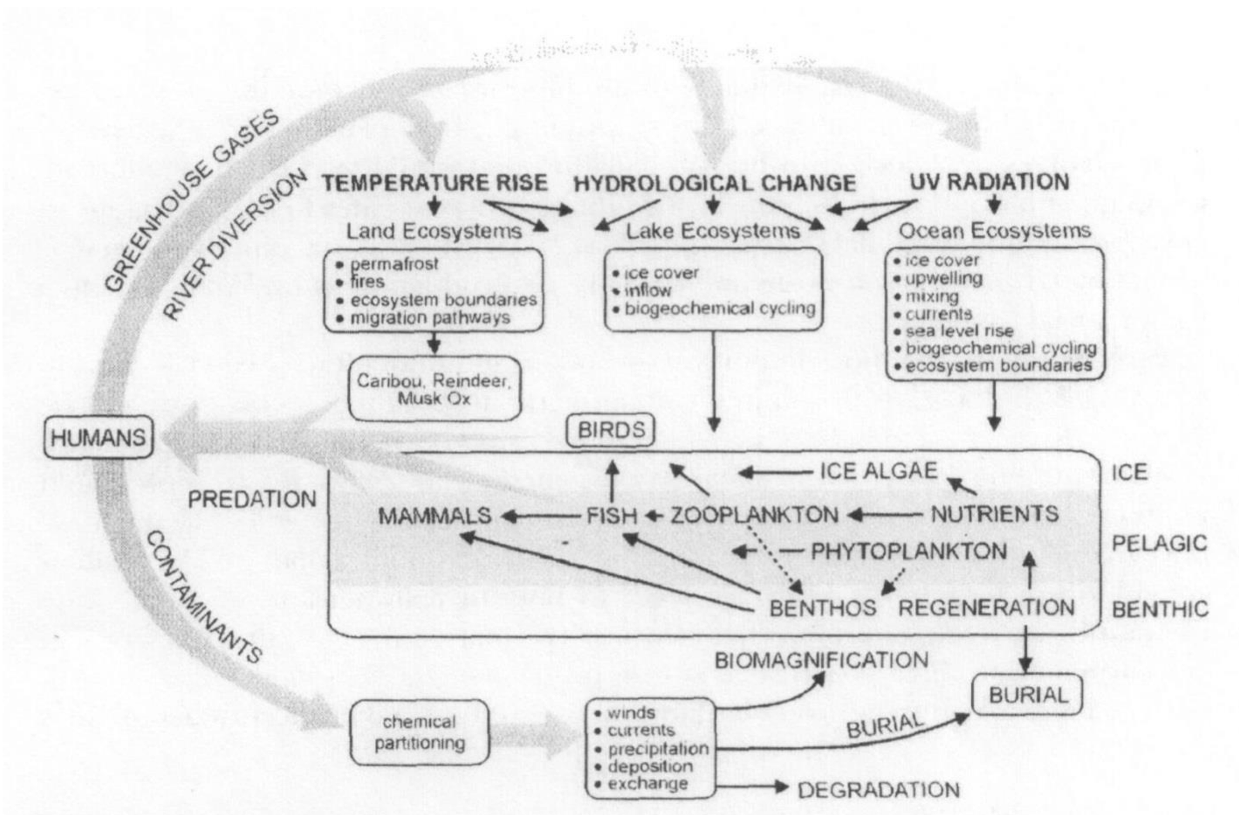
Selenium can be toxic in large concentrations, but trace amounts are actually necessary for cellular function. As mentioned above, in polar bears there is a positive correlation between concentrations of mercury and selenium in liver and other tissues. This correlation may be due to a protective mechanism whereby selenium acts to biotransform methylmercury to a less toxic form. Selenium has also been shown to reduce toxic effects associated with high concentrations of cadmium and silver. Experiments indicate that antioxidants such as reduced glutathione (GSH) and vitamin E also protect against mercury, cadmium, and silver. Therefore, the evidence suggests that selenium may be acting as an antioxidant and forming insoluble complexes with the metals, thus rendering them no longer a toxicity risk (Woshner et al., 2001).

CLIMATE CHANGE

Over the past century, the Earth has warmed 0.6˚C. This is a global average, however, and warming has actually been significantly greater at higher latitudes. During the past four decades, the rate of warming has exceeded that of any time over the past 1,000 years. Ice and snow cover, on average, has decreased 10% since the 1960s (Jenssen, 2006). In 2007, the US Geological Survey estimated a 42% reduction in summer sea ice by 2050. Ice loss of this magnitude would decimate the world's polar bear population, potentially reducing numbers by two-thirds (Courtland, 2008).

Climate change alone poses an imminent threat to the long-term survival of countless species, most notably polar bears (Figure 12).

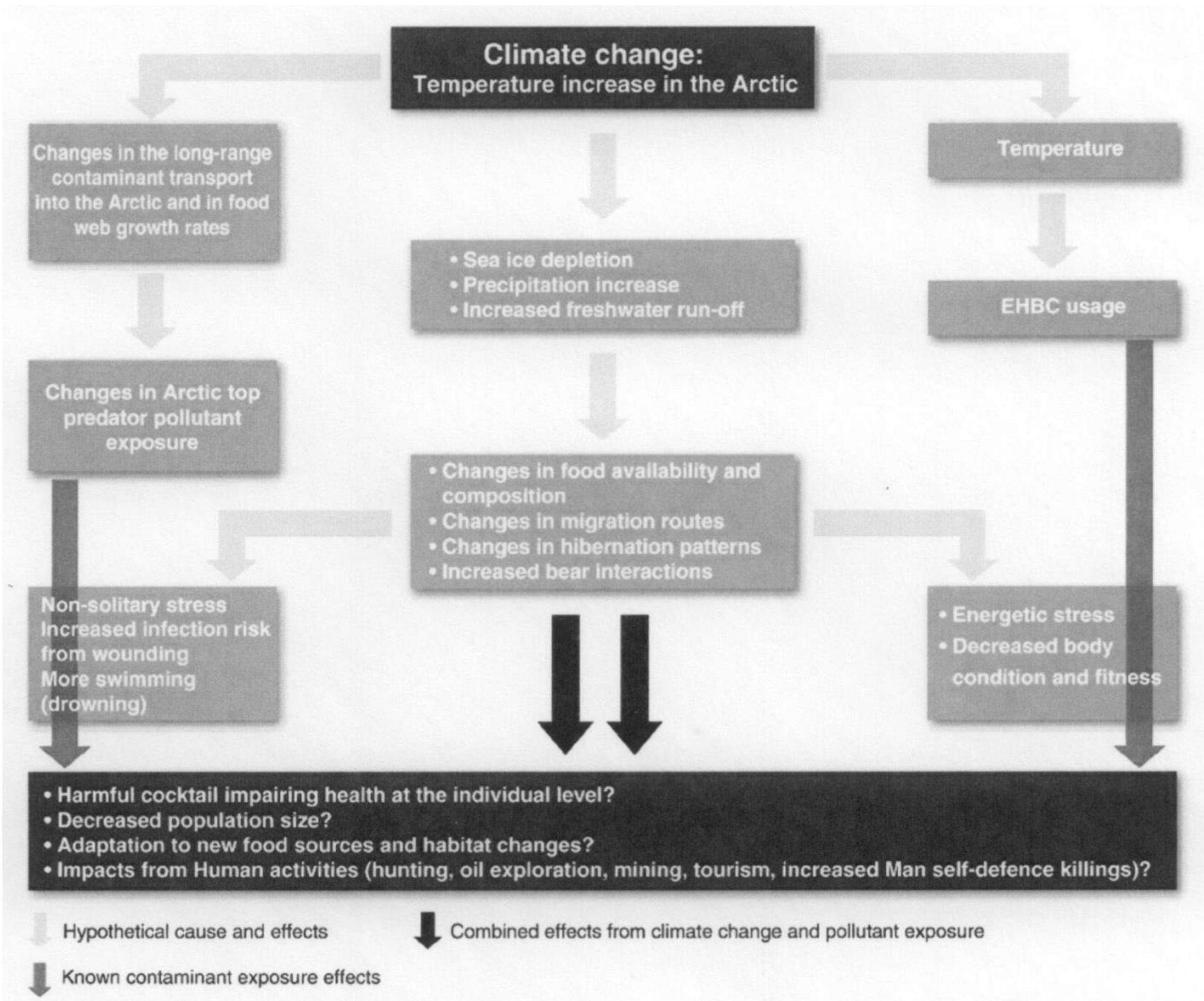
Figure 12. Human activities contributing to global change. (Macdonald et al., 2003)



The southern populations of bears, such as those from western Hudson Bay, will first experience the effects (Prestrud and Stirling, 1994). While increasing temperatures are an issue, the result of the increased temperatures, phase change, is the primary concern in the Arctic. As sea ice melts, polar bears lose habitat and seal hunting becomes more difficult, ultimately leading to nutritional stress, reduced body condition, reduced reproductive rates, increased cub mortality, increased migration distances in search of food, and an increase in bear-human interactions (Macdonald, 2005). Not only will seal hunting become more difficult because of logistical difficulties, seals will be impacted by ice loss and thus experience reduced populations, further complicating successful seal hunting by polar bears (Prestrud and Stirling, 1994). Weight loss will increase contaminant burdens since the contaminants are not reduced in proportion to loss of body fat or muscle. Loss of habitat will lead to population stress, as individuals must increase competition for resources. New pathogens may be introduced in the ballast water of ships which will take advantage of new shipping routes as a result of reduced sea ice. In addition, changes in sea ice cover may result in changes in migration routes, resulting in non-native species entering Arctic waters. These species may be carriers of pathogens not previously found in the Arctic. Due to reduced body condition and heavy contaminant loads, individuals will have poorly functioning immune systems and therefore be susceptible to introduced diseases (Macdonald, 2005). Changes in air and water temperatures and ice cover may attract non-native species into the Arctic or alter the populations of endemic species such that the entire food web is changed. Predators such as polar bears may be exposed to higher contaminant loads or new contaminants by preying on such species.

A study of western Hudson Bay polar bears demonstrated these effects. In this area, the mean annual air temperature has increased 1.5˚C in the past three decades and subsequently the summer sea ice breakup is occurring approximately three weeks earlier, so clearly climate change is already impacting this area. As a result, polar bear access to bearded seals has been reduced. Data indicate that bearded seals possess the lowest contaminant loads of the various seal species in this area. Since access to bearded seals has been reduced, polar bears must shift to consumption of other, more accessible and contaminated, seal species. Not surprisingly, the data indicated a proportional increase in polar bear contaminant burden as a result of the changed diets, thus demonstrating that an altered food web as a result of climate change can increase contaminant load in polar bears (Figure 13) (McKinney et al., 2009).

Figure 13. Combined stress effects from contaminant exposure and climate change. (Sonne, 2010)

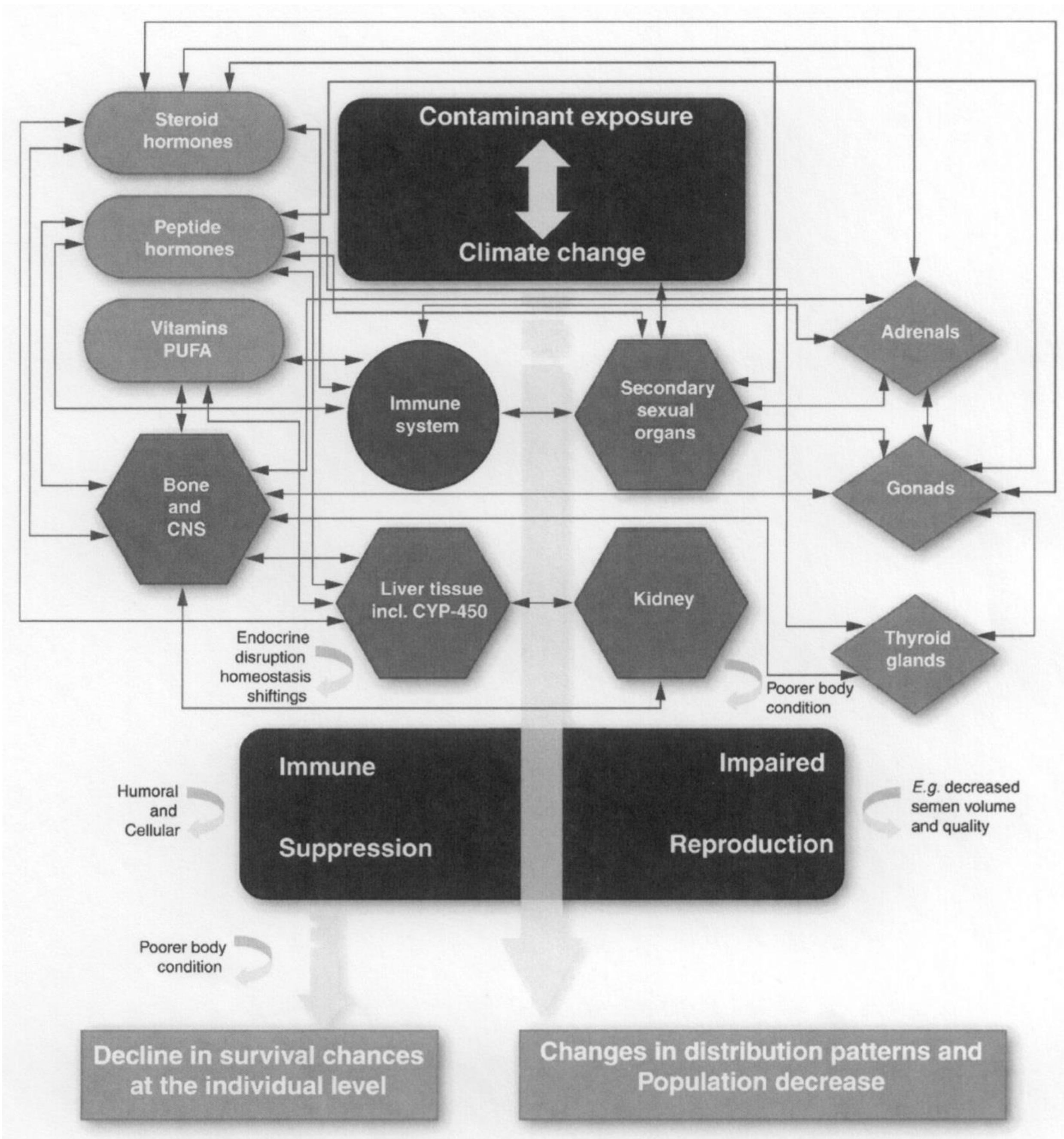


In fact, the combined effects of climate change and contaminants have been demonstrated among these Hudson Bay bear populations as evidenced by data indicating over the past three decades population size has decreased by approximately 22%, subadult survival has decreased, general body condition index has decreased from 0.82 to 0.03 on average, and reproductive success has decreased.

These effects may be mirrored among more northern populations of polar bears as continued warming in the Arctic causes more widespread changes (Sonne, 2010). Elevated temperatures or rainfall during late winter may increase cub mortality due to the collapsing of dens (Prestrud and Stirling, 1994). The changing climate alters the transport, transfer, and capture of contaminants, thus potentially magnifying the negative impacts on Arctic species (Macdonald, 2005). Melting ice can release trapped POPs, heavy metals, and other contaminants which have accumulated for decades in successive layers of ice (Willerroider, 2003). For example, as permafrost melts and flooding regimes of peatlands subsequently change, methyl-mercury exposure will increase concurrently (Macdonald, 2005).

Since the endocrine system is critical for adapting to stressful conditions, the multitude of endocrine disrupting chemicals present in the Arctic will compromise the adaptability of Arctic organisms such as the polar bear to the changing climate and conditions (Jenssen, 2006). This scenario is likely occurring currently among East Greenland and Svalbard populations which are known to possess significant contaminant loads. Over the past 10 years, this region has experienced major sea ice loss. Data indicate a marked decrease in polar bear size among these populations during this period of time, thus suggesting the combined effects of ice loss due to climate change and EDCs resulting in stress and reduced body condition. Ultimately, climate change and the presence of contaminants in the Arctic will synergistically accelerate a reduction in the circumpolar polar bear population (Figure 14) (Sonne, 2010).

Figure 14. Combined stress mechanisms at the individual and population level. (Sonne, 2010)



RECOMMENDATIONS

While an impressive array of data have been thus far amassed with respect to anthropogenic contaminants in the Arctic, there remain many unanswered questions and incomplete data. Data on PCDDs, PCDFs, non-*ortho* PCBs, chlorinated naphthalenes, and chlorinated diphenyl ethers are lacking, especially on potential effects on developing cubs. Most of the research on chlorinated dioxins and furans in the Arctic was conducted in the 1980s and needs to be repeated (Muir et al., 1999). DDT concentrations have been decreasing among polar bear populations since its use was significantly reduced decades ago. However, recently there has been evidence of increasing concentrations among some western North American populations. These increases roughly correspond with the recent reintroduction of DDT for vector control in some countries in Africa and Asia. More research needs to take place to determine if these occurrences are connected, and if so, to what extent (McKinney et al., 2011).

While use of legacy contaminants has been steadily declining, new chemicals are continually increasing in use worldwide. More studies need to be performed on the occurrence and potential biotic impacts of current-use chemicals in the Arctic such as endosulfan, methoxychlor, pentachlorophenol, trifluralin, atrazine, chlorpyrifos, and chlorothalonil, among others (Muir et al., 1999). Current data are insufficient with respect to the sources, pathways, species-specific effects, food web dynamics, and spatial and temporal trends of these chemicals (Braune et al., 2005). The presence of various heavy metals in the Arctic has been established, but distinguishing between natural and anthropogenic sources remains difficult and warrants further study, as do the effects of these metals on Arctic biota.

Temporal trend data for most contaminants are especially limited. Spatial trend data are more comprehensive, but unanswered questions remain. For example, the subpopulation of bears from M'Clure Strait has significantly higher contaminant levels than subpopulations in nearby regions. This anomaly has yet to be explained. In order to successfully model and better predict future contaminant trends, both temporally and spatially, more real-time monitoring of these trends will need to take place in order to generate data for models (Muir et al., 1999).

Given the renal and hepatic lesions present in some populations, the capacity for pathogens to contribute to lesions, and the changing climate which may introduce such pathogens, it would be prudent to conduct research on emerging pathogens among polar bear populations, particularly those in the southern extent of their range (Sonne et al., 2012). Of particular urgency is the need for more research on the effects of climate change in the Arctic, as well as how climate change will influence contaminant exposure, ice conditions, and other factors. Unfortunately, the topic of climate change garners much uninformed debate and speculation among the general public. Yet, most legitimate scientists recognize the reality of the changing climate, even if the precise mechanisms causing the changes are still disputed in some circles.

It is imperative that polar bear biomonitoring and research continue and expand regarding the health and population effects from contaminant exposure and climate change. Furthermore, researchers must make better efforts at communicating results and implications with both the public and the politicians who shape environmental policy. Often, the media is the conduit for this communication, for better or worse. The key to successfully informing politicians and the public is to present findings in terms which are easy to relate and by describing how these impacts will affect the public. In addition, it is important to emphasize the difference between valid science and pseudoscience, as the two are frequently confused in mainstream media and government proceedings.

CONCLUSION

Exposure to contaminants and the effects of climate change such as depletion of sea ice are undoubtedly affecting polar bear health and population size as a result of energetic and physiological stress impacting immune, endocrine, reproductive, and other body systems (Sonne, 2010). Specifically, the East Greenland and Svalbard populations have thus far been most impacted by POPs, while mercury contamination appears to be of particular current biological concern among the Alaskan bear population. Evidence indicates climate change has already significantly affected the Hudson Bay bear population and may be currently impacting other populations as well. Unfortunately, absent drastic international measures to curtail contamination and climate change, these impacts on polar bears will continue to worsen. Future trends will likely mirror current statuses with respect to the distribution and accumulation of POPs and mercury. As climate change progresses, the populations in the Barents, Chukchi, and Beaufort Seas will probably be next to begin to experience significant impacts. Ultimately, all populations will be affected and population decline is essentially a certainty. The questions are: To what extent will the population decline? And is extinction a realistic possibility? Experts in one study estimated a 30% reduction in the total polar bear population by 2050. However, that may be a low estimate, as other research has postulated the population decline by 2050 may be as high as 70%. These numbers are representative of the entire circumpolar population and effects will vary regionally. For example, the same experts who predict a 30% decline in the circumpolar population, specifically predict declines of 63%, 45%, and 38% for the Barents Sea, Hudson Bay, and the Chukchi Sea, respectively. In addition, population declines of 30% and 18% have been estimated by the same experts for the Beaufort Sea and Canadian Archipelago populations, respectively (O'Neill et al., 2008). Experts from another study predict the likely extirpation by 2050 of the populations in Hudson Bay, Foxe Basin, Baffin Bay, Davis Strait, Beaufort Sea, Chukchi Sea, Laptev Sea, Kara Sea, and Barents Sea (Molnár et al., 2010). The Intergovernmental Panel on Climate Change (IPCC) Working Group II has stated that if temperatures warm 2.8°C above pre-industrial levels, polar bears will face a very real threat of becoming entirely extinct (O'Neill et al., 2008). Furthermore, humans are similarly impacted by the very same contaminants and climate change, thus highlighting the urgency of addressing these issues.

While the use of many legacy contaminants has been curtailed internationally, new chemicals are continually being developed and marketed. Just as the historic contaminants such as DDT and others were at the time of their initial use claimed to be "safe," the new chemicals of today benefit from the same advertising claims. Nevertheless, science is proving otherwise.

Contamination in the Arctic is an international problem, since source regions span the globe. Economic and political factors are certainly part of the equation when confronting source reduction, particularly in many developing countries in Asia which are currently the major sources of contaminants reaching the Arctic. A solution will be challenging and require international acknowledgement of the problem and cooperation in reducing source emissions. International bodies and institutions such as the United Nations, World Health Organization, and World Bank will need to become more active in promoting source reduction and proper storage and disposal, particularly in developing countries. Even with effective international measures reducing emissions, due to global distillation, contaminants already in the environment in lower latitudes will continue to be transported to the Arctic for several decades (Bard, 1999). Consequently, threats to polar bears and the Arctic ecosystem will surely be of significant concern for decades to come. When considering the impact of man's actions on polar bears, the Arctic, and indeed the entire biosphere, perhaps John Muir stated it best – "When one tugs at a single thing in nature, he finds it attached to the rest of the world."

REFERENCES

Bard, S.M. 1999. Global transport of anthropogenic contaminants and the consequences for the Arctic marine ecosystem. Marine Pollution Bulletin **38(5)**:356-379.

Bechshøft, T.Ø., Rigét, F.F., Sonne, C., Letcher, R.J., Muir, D.C., Novak, M.A., Henchey, E., Meyer, J.S., Eulaers, I., Jaspers, V.L., Eens, M., Covaci, A., and R. Dietz. 2012. Measuring environmental stress in East Greenland polar bears, 1892-1927 and 1988-2009: what does hair cortisol tell us? Environment International **45**:15-21.

Bechshøft, T.Ø., Sonne, C., Dietz, R., Born, E.W., Muir, D.C., Letcher, R.J., Novak, M.A., Henchey, E., Meyer, J.S., Jenssen, B.M., and G.D. Villanger. 2012. Associations between complex OHC mixtures and thyroid and cortisol hormone levels in East Greenland polar bears. Environmental Research **116**:26-35.

Braune, B.M., Outridge, P.M., Fisk, A.T., Muir, D.C., Helm, P.A., Hobbs, K., Hoekstra, P.F., Kuzyk, Z.A., Kwan, M., Letcher, R.J., Lockhart, W.L., Norstrom, R.J., Stern, G.A., and I. Stirling. 2005. Persistent organic pollutants and mercury in marine biota of the Canadian Arctic: an overview of spatial and temporal trends. Science of the Total Environment **351-352**:4-56.

Brunström, B. and K. Halldin. 2000. Ecotoxicological risk assessment of environmental pollutants in the Arctic. Toxicology Letters **112-113**:111-118.

Bytingsvik, J., Lie, E., Aars, J., Derocher, A.E., Wiig, Ø., and B.M. Jenssen. 2012. PCBs and OH-PCBs in polar bear mother-cub pairs: a comparative study based on plasma levels in 1998 and 2008. Science of the Total Environment **417-418**:117-128.

Chiu, A., Chiu, N., Beaubier, N.T., Beaubier, J., Nalesnik, R., Singh, D., Hill, W.R., Lau, C., and J. Riebow. 2000. Effects and mechanisms of PCB ecotoxicity in food chains: algae → fish → seal → polar bear. Environmental Carcinogenesis and Ecotoxicology Reviews **C18(2)**:127-152.

Corsolini, S., Kannan, K., Imagawa, T., Focardi, S., and J.P. Giesy. 2002. Polychloronaphthalenes and other dioxin-like compounds in Arctic and Antarctic marine food webs. Environmental Science and Technology **36(16)**:3490-3496.

Courtland, R. 2008. Polar bear numbers set to fall. Nature **453(22)**:432-433.

Derocher, A.E., Lunn, N.J., and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology **44(2)**:163-176.

de Wit, C.A., Alaee, M., and D.C. Muir. 2006. Levels and trends of brominated flame retardants in the Arctic. Chemosphere **64**:209-233.

Eaton, R.D. and J.P. Farant. 1982. The polar bear as a biological indicator of the environmental mercury burden. Arctic **35(3)**:422-425.

Fisk, A.T., de Wit, C.A., Wayland, M., Kuzyk, Z.Z., Burgess, N., Letcher, R., Braune, B., Norstrom, R., Blum, S.P., Sandau, C., Lie, E., Larsen, H.J., Skaare, J.U., and D.C. Muir. 2005. An assessment of the toxicological significance of anthropogenic contaminants in Canadian Arctic wildlife. Science of the Total Environment **351-352**:57-93.

Jenssen, B.M. 2006. Endocrine-disrupting chemicals and climate change: a worst-case combination for Arctic marine mammals and seabirds? Environmental Health Perspectives **114(1)**:76-80.

Kucklick, J.R., Struntz, W.D., Becker, P.R., York, G.W., O'Hara, T.M., and J.E. Bohonowych. 2002. Persistent organochlorine pollutants in ringed seals and polar bears collected from northern Alaska. Science of the Total Environment **287**:45-59.

Letcher, R.J., Bustnes, J.O., Dietz, R., Jenssen, B.M., Jørgensen, E.H., Sonne, C., Verreault, J., Vijayan, M.M., and G.W. Gabrielsen. 2010. Exposure and effects assessment of persistent organohalogen contaminants in Arctic wildlife and fish. Science of the Total Environment **408**:2995-3043.

Macdonald, R.W. 2005. Climate change, risks and contaminants: a perspective from studying the Arctic. Human and Ecological Risk Assessment **11**:1099-1104.

Macdonald, R.W., Mackay, D., Li, Y.F., and B. Hickie. 2003. How will global climate change affect risks from long-range transport of persistent organic pollutants? Human and Ecological Risk Assessment **9(3)**:643-660.

McKinney, M.A., Letcher, R.J., Aars, J., Born, E.W., Branigan, M., Dietz, R., Evans, T.J., Gabrielsen, G.W., Peacock, E., and C. Sonne. 2011. Flame retardants and legacy contaminants in polar bears from Alaska, Canada, East Greenland and Svalbard, 2005-2008. Environment International **37**:365-374.

McKinney, M.A., Peacock, E., and R.J. Letcher. 2009. Sea ice-associated diet change increases the levels of chlorinated and brominated contaminants in polar bears. Environmental Science and Technology **43(12)**:4334-4339.

Molnár, P.K., Derocher, A.E., Thiemann, G.W., and M.A. Lewis. 2010. Predicting survival, reproduction and abundance of polar bears under climate change. Biological Conservation **143**:1612-1622.

Muir, D., Braune, B., DeMarch, B., Norstrom, R., Wagemann, R., Lockhart, L., Hargrave, B., Bright, D., Addison, R., Payne, J., and K. Reimer. 1999. Spatial and temporal trends and effects of contaminants in the Canadian Arctic marine ecosystem: a review. Science of the Total Environment **230**:83-144.

Muir, D.C., Wagemann, R., Hargrave, B.T., Thomas, D.J., Peakall, D.B., and R.J. Norstrom. 1992. Arctic marine ecosystem contamination. Science of the Total Environment **122**:75-134.

O'Neill, S.J., Osborn, T.J., Hulme, M., Lorenzoni, I., and A.R. Watkinson. 2008. Using expert knowledge to assess uncertainties in future polar bear populations under climate change. Journal of Applied Ecology **45**:1649-1659.

Oskam, I.C., Ropstad, E., Lie, E., Derocher, A.E., Wiig, Ø., Dahl, E., Larsen, S., and J.U. Skaare. 2004. Organochlorines affect the steroid hormone cortisol in free-ranging polar bears (*Ursus maritimus*) at Svalbard, Norway. Journal of Toxicology and Environmental Health **67**:959-977.

Prestrud, P. and I. Stirling. 1994. The international polar bear agreement and the current status of polar bear conservation. Aquatic Mammals **20(3)**:113-124.

Ropstad, E., Oskam, I.C., Lyche, J.L., Larsen, H.J., Lie, E., Haave, M., Dahl, E., Wiger, R., and J.U. Skaare. 2006. Endocrine disruption induced by organochlorines (OCs): field studies and experimental models. Journal of Toxicology and Environmental Health **69**:53-76.

Skaare, J.U., Larsen, H.J., Lie, E., Bernhoft, A., Derocher, A.E., Norstrom, R., Ropstad, E., Lunn, N.F., and Ø. Wiig. 2002. Ecological risk assessment of persistent organic pollutants in the Arctic. Toxicology **181-182**:193-197.

Sonne, C. 2010. Health effects from long-range transported contaminants in Arctic top predators: an integrated review based on studies of polar bears and relevant model species. Environment International **36**:461-491.

Sonne, C., Letcher, R.J., Leifsson, P.S., Rigét, F.F., Bechshøft, T.Ø., Bossi, R., Asmund, G., and R. Dietz. 2012. Temporal monitoring of liver and kidney lesions in contaminated East Greenland polar bears (*Ursus maritimus*) during 1999-2010. Environment International **48**:143-149.

Tenenbaum, D.J. 2004. POPs in polar bears: organochlorines affect bone density. Environmental Health Perspectives **112(17)**:1011.

Verreault, J., Muir, D.C., Norstrom, R.J., Stirling, I., Fisk, A.T., Gabrielsen, G.W., Derocher, A.E., Evans, T.J., Dietz, R., Sonne, C., Sandala, G.M., Gebbink, W., Riget, F.F., Born, E.W., Taylor, M.K., Nagy, J., and R.J. Letcher. 2005. Chlorinated hydrocarbon contaminants and metabolites in polar bears (*Ursus maritimus*) from Alaska, Canada, East Greenland, and Svalbard: 1996-2002. Science of the Total Environment **351-352**:369-390.

Willerroider, M. 2003. Roaming polar bears reveal Arctic role of pollutants. Nature **426(6)**:5.

Woshner, V.M., O'Hara, T.M., Bratton, G.R., and V.R. Beasley. 2001. Concentrations and interactions of selected essential and non-essential elements in ringed seals and polar bears of Arctic Alaska. Journal of Wildlife Diseases **37(4)**:711-721.

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